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LISTOWEL ARTIFICIAL MARSH

PROJECT REPORT

October, 1986

Ontario

Ministry of
Environment
and Energy

LISTOWEL ARTIFICIAL MARSH PROJECT REPORT

By

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Prepared for
Research Advisory Committee
Ontario Ministry of the Environment
Project No. 128 RR

October 1986

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ACKNOWLEDGEMENTS

This document is the result of the cooperation of many people. We would like to express our appreciation to S.A. Black, I. Wile and G. Miller of the former Pollution Control Branch and the Water Resources Branch of the Ministry of the Environment who were involved in the planning and development of the Listowel project. We would also like to thank the following people for their contributions to the study.

- W. Lewandowski and D. Snell, Water Resources Branch, MOE
- G. Palmateer and staff, Southwestern Region, MOE
- G. Hutchin, N. Carr and M. Seguin, Town of Listowel
- P. Pos and G.A. Surgenor, University of Guelph.

Funding for the study provided by the Research Advisory Committee of the Ministry of the Environment is gratefully acknowledged.

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SUMMARY

Marsh treatment of municipal sewage has received attention in recent years as a cost-effective alternative to lagoons in rural communities and small urban centres. The Listowel Artificial Marsh Project was conducted to investigate the efficiency and feasibility of year-round marshland treatment of wastewater and to provide guidelines for the design and operation of these systems in Ontario. In addition, the study examined the role of various marsh components (e.g. growing plants, litter, sediment bacteria) in wastewater renovation and the fate of wastewater constituents.

An experimental facility was constructed adjacent to the existing sewage works in the Town of Listowel and was operated for four years (1980-84). Sewage treatment at Listowel consists of alum injection prior to an aeration cell which is followed by two wastewater stabilization lagoons operated in series. The pilot project consisted of five separate experimental marsh treatment systems which together treated 4% of the town's sewage (generated at the average rate of 1.2 MIGD).

Several marsh system designs and types of pre-treatment were tested. The designs included a channelized marsh with serpentine configuration, a shallow marsh and a complex of shallow marsh, deep pond and channelized marsh. Pre-treatment types (diverted from the existing sewage system) included 1) aerated cell effluent and 2) lagoon effluent. Operation of the marsh systems allowed flexibility in hydraulic loading rates, water depths and detention times.

Cattails (*Typha* spp.) were easily propagated in the marsh systems and produced a high biomass under permanently flooded conditions. Harvesting (including years with multiple harvests) removed a small part (<10%) of the total annual nitrogen and phosphorus input and is not considered cost-effective. However, nutrient assimilation by marsh plants takes place during the growing season when receiving waters are most sensitive. During this period, nitrogen and phosphorus were also shown to be immobilized by bacteria involved in litter decomposition.

The recommended sewage pre-treatment facility is an aerated facultative cell with a 30 day hydraulic retention time and phosphorus reduction to 1.0 mg/L TP. These provide necessary phosphorus, BOD_5 and suspended solids reduction prior to marsh treatment, as well as sufficient sludge storage capacity for a design period of 20 years. The channelized marsh configuration showed greater treatment efficiencies than the other marsh designs tested. Hydraulic retention time in the marshes was effectively regulated by adjustments of water depth ($\leq$10 cm in summer, $\geq$30 cm in winter) to compensate for seasonal influences, such as evapotranspiration and ice formation. The data indicated a reduction in treatment efficiency when the hydraulic detention time in the marsh systems deviated from the range of 7-14 days.

Wastewater renovation of suspended solids, BOD_5 and total phosphorus achieved in the marsh systems was at levels between secondary and tertiary treatment. The four year annual average effluent quality in the channelized marshes in terms of BOD_5, SS, TP and TKN was 7.6, 9.2, 0.5 and 6.2 mg/L, respectively, in the system receiving lagoon effluent (System 3) and 9.6, 8.0, 0.6 and 8.7 mg/L, respectively, in the system receiving aerated cell effluent (System 4).

The data suggest saturation of the soluble phosphorus adsorption capacity of the sediment in the Listowel marsh systems. Periodic oxygen depletion and high phosphorus loadings during the first year may have contributed to the reduction in soluble phosphorus retention. The possible recharge of the sediment phosphorus retention capacity by clay and silt particles and their associated mineral component was precluded by the absence of stormwater in the influent. The soluble phosphorus retention capacity was inversely related to cumulative soluble phosphorus loadings. Annual total phosphorus removal efficiencies remained relatively stable throughout the four years.

Elevated ammonia levels were coincidental with oxygen depletion in summer and low temperatures in combination with low oxygen levels in winter. Hydrogen sulfide and phenol concentrations increased in some years during periods when conditions were conducive to anaerobic microbial metabolism. Large reductions in fecal bacterial indicators were achieved during most of the year, however, increases in summer and/or winter often exceeded Ministry objectives for disinfected secondary effluent. Bacterial pathogens were generally detected in the marsh effluents at low levels but increases were detected in some winter months.

The recommended sewage pre-treatment process, which has been incorporated into the design of a full-scale marsh treatment facility at Port Perry, Ontario, ensures low levels of influent solids and phosphorus, as well as consistently high dissolved oxygen concentrations in wastewater entering the marsh. Successful utilization of the marshland process in southern Ontario is predicated on the production of an effluent quality high enough to permit year-round discharge. Since many receiving waters have extremely low flows and consequently low assimilative capacities during the summer months, a very high effluent quality is essential. The Port Perry marsh treatment facility will be monitored to establish the treatment capabilities of the process and to determine the potential for application in southern Ontario.

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1.0 INTRODUCTION

The use of marshes for wastewater treatment has received attention in recent years due to reports indicating that they may provide a viable alternative to conventional practices of wastewater treatment in rural communities and small urban centres. In Ontario, seasonal retention lagoons are common in many of these communities. In most cases, the lagoons must be large enough to retain the wastewater from 3-12 months, until there is adequate dilution in the receiving waters to meet Ministry Water Quality Guidelines. Marsh treatment has the advantage of being less land consumptive than seasonal retention lagoon systems, more efficient than continuous discharge lagoons, and considerably more economical and energy efficient than mechanical systems.

Studies of marsh wastewater treatment have largely focused on the use of natural wetlands for tertiary treatment. Artificial marshes, however, may offer greater scope for general use and are not restricted by many of the environmental concerns and user conflicts associated with natural wetlands. Unlike natural marshes which are confined by the proximity to the sewage source, artificial marshes can be constructed anywhere, including sites with limited alternative uses. A major advantage of artificial marshes is the control over hydraulic pathways and retention time, both of which have been shown to affect effluent quality.

In order to assess the potential of artificial marshes for year-round sewage treatment in Ontario, an experimental facility was constructed adjacent to the existing sewage works in the Town of Listowel. It was anticipated that regular monitoring of the facility would yield data on the efficiency and feasibility of marshland wastewater treatment.

2.0 OBJECTIVES

The primary objectives of the Listowel Artificial Marsh Project were:

a) to determine the capacity of artificial marshes to renovate wastewater on a year-round basis in Southern Ontario,

b) to define guidelines for the design and operation of marshland wastewater treatment systems, such as the degree of pre-treatment required prior to waste discharge to the marsh, loading rates, retention times and water depths.

Several secondary objectives were related to the fundamental understanding of marsh treatment systems and to the development of non-routine operational procedures. These included:

c) an assessment of the role of various marsh components in the removal and ultimate fate of wastewater constituents,

d) an evaluation of the impact of vegetation harvesting on effluent quality and an assessment of potential uses of the harvested biomass,

e) an investigation of nuisance organisms generated within the marsh environment, their significance and possible mitigative measures.

3.0 DESCRIPTION OF STUDY SITE AND FACILITIES

The Town of Listowel operates waste stabilization ponds which treat sewage at the rate of 1.2 MIGD. Approximately 35% of the total sewage volume treated is generated by the municipality. The remaining flow, including 70% of the BOD loadings, is generated by a food processing plant which includes poultry handling facilities. A 0.7 ha (1.8 acre) aeration cell (Cell A) with 3.5 days retention is followed by two facultative lagoons (Cells B + C) with a combined 85 day retention capacity (Figure 3.1). The lagoons are operated in series and together occupy 28 hectares (69 acres). Effluent discharge is continuous in winter. The lagoons are drawn down in early spring and, from approximately May through October, the contents are spray irrigated on adjacent land. Alum additions of 50 ppm were administered to the wastewater continuously prior to entry into the aeration cell in all but the summer months. However, following the summer of 1982, alum was added to the wastewater throughout the year at the same rate of application.

The artificial marsh systems were constructed adjacent to the Listowel lagoons and commenced operation in August, 1980. The marsh project consisted of five separate experimental treatment systems (Figure 3.2, Table 3.1) which together received approximately 4% of the town's sewage. System 1 was composed of three cells operated in series: a shallow marsh (2120 m^2), a pond (728 m^2) and a channelized marsh with serpentine configuration (1324 m^2). The channels were 4 m wide and totalled 334 m in length (length width ratio 84:1 and 17:1 in each channel). The earthen berms of the channelized marsh were designed to accommodate a tractor with a cutting bar for harvesting the cattails. Systems 3 and 4 were channelized marshes of the same size (1324 m^2) and design as the third cell of System 1. System 2 (909 m^2) and 5 (941 m^2) were non-channelized shallow marshes. Systems 1, 2 and 3 received wastewater effluent from the first of the two Listowel lagoons (Cell B) whereas Systems 4 and 5 received alum-treated effluent from the existing aeration cell (Cell A).

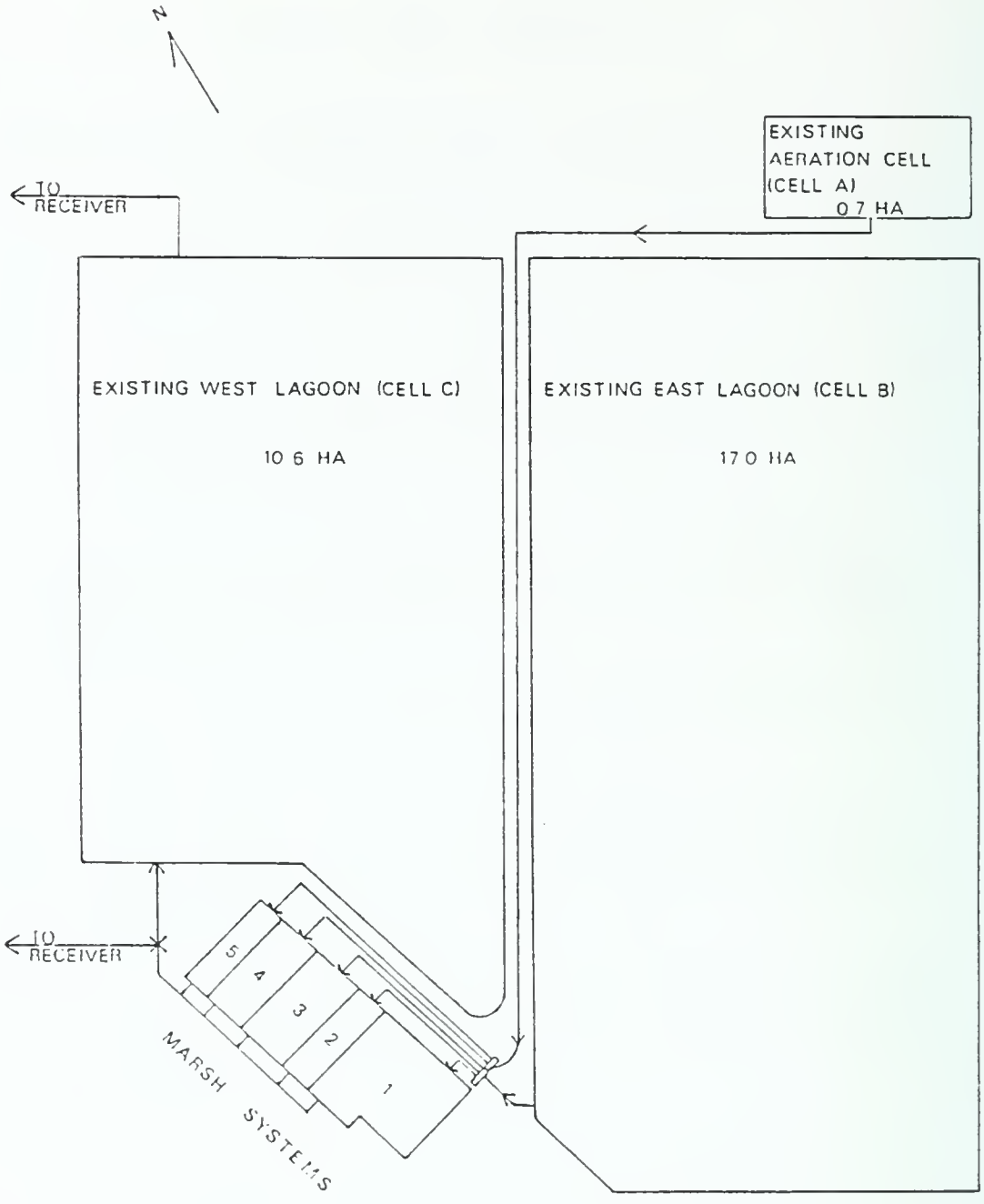


Figure 3.1 Location of artificial marsh systems with respect to existing Town of Listowel sewage treatment facilities.

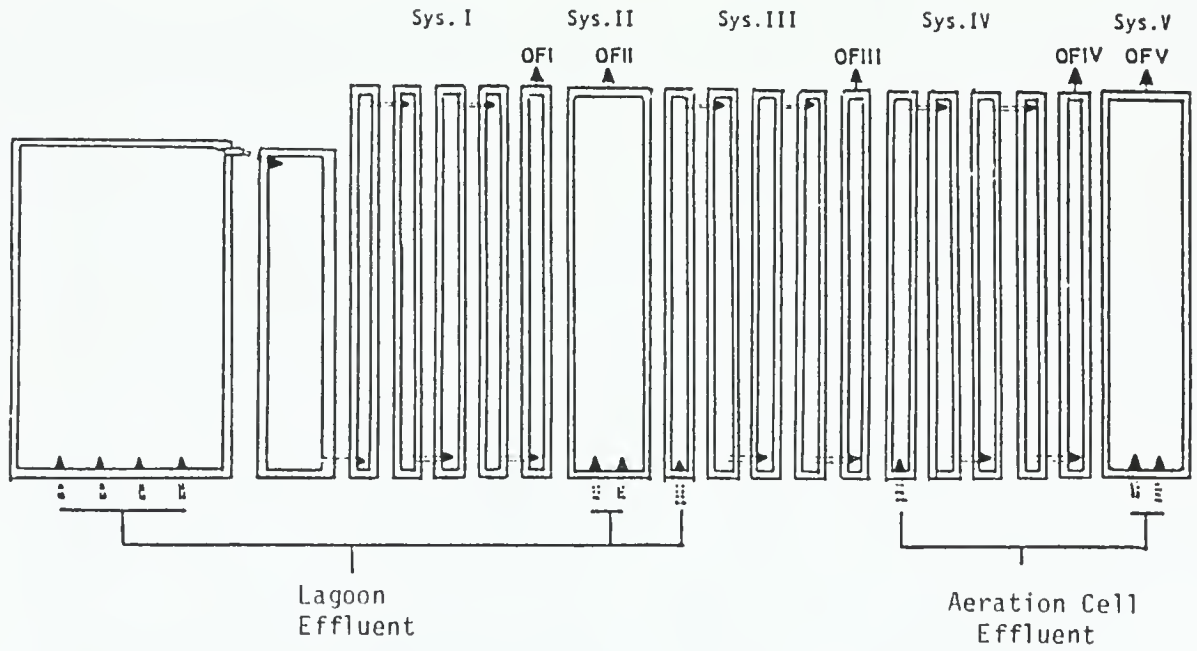


Figure 3.2 Design of Listowel marsh systems.

Table 3.1 Description of Marsh Systems and Types of Pre-treatment

System	Description	Surface area (m^2)	Drainage basin area (m^2)*	Type of pre-treatment
1	large marsh-	2,120	591	aeration cell (w/alum) + lagoon
	deep pond-	728	389	
	channelized marsh	1,324	1,515	
		TOTAL 4,172	TOTAL 2,495	
2	shallow marsh	909	445	aeration cell (w/alum) + lagoon
3	channelized marsh	1,324	1,442	aeration cell (w/alum) + lagoon
4	channelized marsh	1,324	1,442	aeration cell (w/alum)
5	shallow marsh	941	522	aeration cell (w/alum)

* Derived from the area occupied by a 1.5 metres strip around marsh systems (plus berm area in Systems 1, 3 and 4).

All systems were provided with a basin of compacted clay (6 inches) to prevent infiltration of ground water and exfiltration of wastewater. The marsh systems were backfilled with 3 inches of top soil and topped-off with an additional 3 inches of a mixture of top soil and peat (10% by volume) to provide an adequate growth medium for cattails. Laser grading with an accuracy of 5 cm was employed to permit the even distribution of the flow.

Cattails (Typha spp.) were planted at 1 m intervals in mid-July, 1980 in all systems (except the pond cell of System 1) and were fully established by mid-August of the same year. The species composition was predominantly Typha latifolia with some T. angustifolia present. Submerged aquatic plants, Potamogeton spp. and Elodea canadensis, were planted in the pond of System 1. The berms and surrounding land was seeded with a mixture of rye grass, Kentucky blue grass, red fescue and clover to prevent erosion on the 2:1 slope to the cells.

Wastewater from the existing Listowel lagoon (Cell B) and aeration cell was pumped to two flow splitting boxes (SBLE - splitter box with lagoon effluent, SBAS - splitter box with aeration cell effluent) and then fed by gravity to the experimental marsh systems. Inflows and outflows were monitored by v-notch weirs and water level recorders in order to obtain mass balances of wastewater constituents. The 60^{o} V-notch weirs were replaced with 30^{o} V-notch weirs in May 1982 when hydraulic loadings were reduced. The operational water depths (5-34 cm) were controlled by stop logs located at the terminal point in each system with the exception of the pond in System 1 which was maintained at a depth of 90 cm. The discharge from all systems was collected in a wet well and, depending on marsh effluent quality, either carried by gravity to the existing Chapman Drain which flows into the Middle Maitland River or pumped into the second Listowel lagoon (Cell C).

An instrumental meteorological station was set up on site to monitor precipitation and total nutrient input from atmospheric disposition (during both wet and dry weather).

Airblowers with perforated PVC lines were inserted in each of the connecting culverts of the channelized marshes in winter to reduce chances of flow blockage due to ice accumulation. Small propane heaters were installed in the insulated effluent chambers to prevent ice build-up in the weirs.

4.0 MONITORING PROGRAM

4.1 Influent and Effluent Water Quality

Sampling stations were established inside the splitter box and at all outflow weirs to determine concentrations of wastewater constituents entering and exiting each marsh system. Routine parameters, including pH, dissolved oxygen (DO), conductivity, water temperature, suspended solids (SS), biological oxygen demand (BOD_5), total phosphorus (TP), soluble reactive phosphorus (SRP), total Kjeldahl nitrogen (TKN), ammonia (NH_3), nitrate plus nitrite (NO_{2+3}), chlorophylls a + b and fecal indicator bacteria were monitored weekly from April to November and bi-weekly from December to March. All water samples were collected between 8 a.m. and 9 a.m.

Hydrogen sulfide and phenol levels were analysed most intensively in winter. Two to three samples were tested for these parameters during the months of December through March. In some other months tests were conducted on one occasion. Precipitation collected on site was analyzed for TP, TKN, NO_{2+3} and NH_3. Analytical techniques are described elsewhere (MOE, 1981).

Un-ionized ammonia concentrations were calculated from an equation using temperature and pH as independent variables (Emerson et al, 1975). Levels of un-ionized hydrogen sulfide were determined by interpolation from a table of undissociated H_2S concentrations at various pH values and temperatures in a solution of ionic strength 0.0033 (Clarke, 1974) . The latter is comparable to the ionic strength of domestic wastewater.

Water samples were analyzed for four pathogenic bacteria commonly found in domestic sewage: Yersinia enterocolitica, Clostridium perfringens, Pseudomonas aeruginosa and Salmonella sp. The methods of recovery, isolation and identification are described elsewhere (Palmateer et al, 1984). Tests were conducted six times per year for Salmonella sp. and primarily during the winter months for the other bacterial pathogens.

Monthly bacterial levels presented in this report are geometric mean concentrations. The geometric mean has the advantage of not being unduly influenced by extreme values in a distribution and is more typically representative of the distribution then the arithmetic mean. The geometric mean was calculated by taking the n^{th} root of the product of n bacterial samples.

4.2 Channel Studies

The pattern of contaminant removal in the channelized systems (3 & 4) was examined by channel sampling undertaken from May, 1983 to September, 1984. Sampling stations were located in the culverts at the end of Channels B through E. When combined with samples taken from the system inflow (preceding channel A) and outflow, it was possible to get an indication of where changes in the water quality parameters were occurring in the channelized systems. Samples were analyzed for SS, BOD_5, TP, TKN, NH_3 and NO_{2+3}. Sampling frequency was the same as for the routine parameters sampled in each system as described above. To best represent the influent quality of wastewater undergoing treatment in the channels, the mean influent concentration for the preceding 4 weeks was used in preparing the graphs.

The oxidation reduction (redox) potential in the top layer of sediment was measured in the channelized systems in an effort to understand the biochemical processes involved in the cycling of nutrients in marsh systems. Observations of a release of phosphorus from the sediment as redox potentials decrease (Wetzel, 1975) have relevance for marshland wastewater treatment of this nutrient.

Nine test stations were located in 3 channels of Systems 3 and 4 during the ice-free period of 1983 and 1984. The stations were sited at the beginning, middle and end of the channels as follows: Channel A (Stations 1, 2 & 3), Channel C (Stations 4, 5 & 6), Channel E (Stations 7, 8 & 9).

Redox potentials were measured with an Orion platinum electrode in conjunction with a Radiometer field pH-Eh meter (Model 29). The electrode was fitted to accommodate a plastic weighted cap with a lip so that all readings were taken at a constant 1.5 cm depth. The filling solution for the electrode was 4 M KCL saturated with silver chloride (90-00-11) which is recommended when the ionic strength is above 0.2 M. Corrections were applied for temperature and pH so that redox potentials could be compared between sites.

4.3 Sediment Bacteria

Sediment samples were collected from Systems 2, 4 and 5 in 1981 and 1982 at 4 stations in each system. Two stations were located one third distance from the front end and two stations at one third distance from the effluent end of each system. Three sediment cores, penetrating to a depth of 2 cm, were retrieved at each station. Samples were collected in alternate months in 1981 and 4 times during 1982. Sediment redox potentials were monitored at each station (at 2 cm depth) when samples were collected. Sediment samples were analyzed for the following bacteria: nitrifiers, Nitrosomonas and Nitrobacter, denitrifiers and sulfate-reducers. Bacterial analyses were performed in accordance with Ministry procedures (MOE, 1981). The analyses were conducted and sediment bacteria graphs were prepared at the London regional office, MOE.

4.4 Hydrology

The hydrology of each marsh system was monitored to provide data for mass loadings and estimates of nutrient removal efficiencies. Major hydraulic inputs to each system included the wastewater inflow, precipitation and terrestrial runoff; outputs included the wastewater outflow and losses through evapotranspiration. Wastewater inflow and outflow sites were equipped with 60^{o} V-notch weirs (until May 1982) or 30^{o} V-notch weirs (May 1982 onward) and stilling wells. Levels in the stilling wells were recorded with Leopold and Stevens Type A Model 71 float-operated recorders. Stage-discharge curves were established for each weir. Instantaneous discharge measurements were taken weekly (and during periods of high and low flow) by collecting the water flowing through the weir in a bucket during a measured period of time. An average water volume from 3 bucket tests was used to derive an estimate of the flow rate in L/sec. Continuous stage records were converted to continuous discharge using a digitizer and STREAM programme (Environment Canada, 1977). These data were subsequently converted to average daily flows for each weir.

Measurements of precipitation were recorded daily, except on weekends, at the on-site meteorological station. Rain depth was measured with a standard 10 cm diameter rain gauge and snow depth was calculated using a Standard Atmospheric Environment Service (A.E.S.) nipher collector. On rare occasions when data were missing, daily precipitation estimates were taken from the nearest A.E.S. meteorological station at Mount Forest. Water loading to the marsh from precipitation was calculated by multiplying precipitation depth by marsh surface area.

Evaporation data were used as an estimate of water losses due to evapotranspiration in the marshes. Although controversial, some recent work suggests that when vegetation always has sufficient water to draw on (as in irrigation ditches or marshes), potential evapotranspiration is approximately equal to "free-water" evaporation from small lakes since both are limited by energy available from heat exchanges with the atmosphere (Bruce and Weisman, 1966). Other studies indicate that marsh system evapotranspiration is greater than evaporation from open water (Brezny et al, 1973; Smid, 1975). However, the estimates of summer evapotranspiration used in this report (e.g. 1980/81 and 1981/82 average of .52 cm/d, range .42 - .64 cm/d) are similar to published marsh evapotranspiration values derived by direct measurement (Zoltek et al, 1978) and calculation using temperature and water vapor pressure (Smid, 1975).

Computed lake evaporation values (Kohler et al, 1975) derived from pan evaporation data measured at the Elora Research Station (40 km east of Listowel) were used to estimate monthly evapotranspiration rates (Evaporation Abstract, 1970-1983, A.E.S.). Lake evaporation values were used because the heat exchange in a marsh is similar to that of a small lake. Evaporation was assumed to be zero from November 1 to March 31. Missing monthly values were replaced by a ten year average for that month (except for the month of April for which only 3 years of data between 1970 and 1983 were available).

Terrestial drainage entering the marsh systems was estimated by using 40% of the precipitation falling in the drainage basin. This estimate was derived from the findings of the Lakeshore Capacity Study for the watersheds of the Muskoka-Haliburton region (Scheider et al, 1983). Because the land surrounding the Listowel artificial marsh systems is extremely flat, the estimate of the drainage area, which includes the berms and a 1.5 m strip of land bordering each system, probably represents a maximum value. Runoff was calculated by multiplying the precipitation depth (X.40) by the estimated drainage area.

An examination of the combination of estimated and measured hydrologic inputs and outputs in the marsh systems indicated frequent seasonal net losses during the four years. The data suggested errors (underestimates) in the outflow discharge measurements rather than exfiltration as the monthly net losses were inversely related to outflow volume (see Section 5.1.2). Outflow discharge rates were often low, particularly in summer due to evapotranspiration. The v-notch weirs were designed to handle higher flow volumes so that often the flows were at the lower limit of the notch capacity resulting in flow measurement error. A recommended minimum head of 6 cm was generally achieved only on infrequent occasions (e.g. during rainfall events).

4.5 Mass Balances

Comparisons of concentrations of wastewater constituents in the influent and effluent can be misleading due to the concentrating effect of evapotranspiration and ice formation and dilution from rainfall. Therefore, to determine the removal efficiency of the marsh systems, mass balances were calculated. Substance loads (g/m^2) into and out of the marshes were derived from daily flows and concentrations measured weekly (biweekly during winter months) at both the inflow and outflow sites. The concentrations were multiplied by the flow volumes for the period midway to the preceding concentration measurement and midway to the next concentration measurement.

Due to the inability to account for all the hydrologic input by the outflow discharge measurements plus estimates of evapotranspiration, adjustments were made to the water balance based on the worst case scenario, i.e., that the outflow volume was underestimated. Although some of the flow measurement error may have occurred at the inflow weir, adjustments were made to the outflow discharges assuming all error from that source. In months where the hydrology

indicated that total hydrologic input exceeded total output, the outflow volume and consequently the effluent loads were raised. The monthly outflow volume was multiplied by the ratio of monthly total hydrologic input over monthly total hydrologic output. The mass balance graphs with seasonal removal efficiencies and all references to loads and treatment efficiencies in this report are based on the effluent loads adjusted as described above.

Seasons were defined with regard to both hydrology and temperature as follows: fall (Sept-Nov.), winter (Dec.-Feb.), spring (Mar.-May) and summer (June-Aug.). For each substance of concern the input sources included in the calculation were the wastewater influent and atmospheric deposition on the marsh surface. The only loss considered here was through the outflow. The difference between total substance input and loss through the marsh effluent (i.e. output) was assumed to represent the efficiency of the marsh system to remove, transform or retain selected wastewater constituents.

4.6 Retention Time

The theoretical retention time in the marsh systems was calculated by dividing the plot volume (m^3) by the inflow rate (m^3/d). Plot volume was derived from the marsh surface area multiplied by water depth, less 5% for the space occupied by the submerged parts of the cattail plants and litter from previous year. Sampling in 1983 indicated that 2% of the volume was occupied by the cattail standing crop. The theoretical residence time does not take into account the environmental factors such as precipitation and evapotranspiration which alter the total water volume in the marsh. The effect of ice formation in reducing retention time is taken into consideration by using measurements of the water depth under the ice cover, although some ice blockages may have gone undetected.

Dye studies were conducted to determine actual retention times. Injections of 20% Rhodamine W.T. dye were made at the splitter box. Water samples, collected with an automatic liquid sampler at 3 hour intervals at the outflow were measured for relative fluorescence with a Turner fluorometer (Model 111).

Relative dye intensity in the samples was plotted against time. The hydraulic residence time from the tracer response curve was assumed to be equal to the centroid of the area under the tracer response curve. The following formula was used to calculate retention time:

$$\text{retention time} = t + y$$

$$t = \frac{\Sigma\ (A \times d)}{\Sigma\ A}$$

Where: A = area under curve per time interval
d = cumulative time interval from start of dye flow to centroid of curve.
y = time from injection to first detection of dye in effluent.

4.7 Biomass Production and Harvesting

In the channelized systems, cattail shoots were cut just above the water level with a tractor-mounted harvester designed for this purpose by the Engineering Faculty at the University of Guelph. The harvester was operated from the berms separating the channels and consisted of a cutting bar and conveyor to transport cut stalks to the berm.

The cutting bar extended above the channel surface, leaving the sediment undisturbed. The fresh weight of the harvested plant material was determined on site and subsamples were collected for determinations of the percent dry weight and nutrient content (TKN and TP) of the dried samples. The amount of nitrogen and phosphorus (g/m^2) removed from the marsh by harvesting was calculated from these data. Multiple harvests were compared with single harvests in their effectiveness in removing nutrients from the marsh systems.

A secondary use for the harvested plant biomass, the production of commercial fuel, was investigated by the Engineering Faculty at the University of Guelph. The study, which was conducted in the summer of 1982, evaluated cattails as a substrate in anaerobic digesters for the production of biogas.

Harvested cattail shoots from System 1 (Channel E) were collected on 2 occasions (in Aug. and Sept. 1982). The plant material was chopped and liquified in a Waring blender and loaded into a 68L bench-scale digester designed for the study. In addition, harvested cattail shoots and whole plants from the artificial marsh, as well as whole plants from a natural marsh, were separated into plant parts (i.e. rhizomes, leaves, stems, seed pods). These were weighted and analyzed for inorganic and organic constituents before and after undergoing digestion. The analyses yielded data on the distribution of fermentable substances in cattails and the composition of the digester effluent. Gas yield was monitored by wet-test gas meter and the gas composition was determined periodically. The digester and methods used in the study are described elsewhere (Pos, 1983).

4.8 Litter Component

In order to gain a better understanding of the fate of nutrients taken up by marsh plants during the growing season, an investigation into plant litter decomposition was undertaken at a natural marsh in Bradford, Ontario. Suspended and submerged cattail shoots were studied for a 2 year period (Oct. 1979 to Oct. 1981) to determine the rate of decomposition and patterns of nutrient gains and losses following senescence of the plants. The method used was adapted from similar work done by Mason and Bryant (1975) and Davis and van der Valk (1978).

Cattail shoot material, collected in early October, was placed in nylon mesh bags with .70 mm aperture diameter, large enough to permit entry to small invertebrates. Sixty bags were suspended and the same number were submerged in the marsh. Three suspended and three submerged bags were retrieved for analyses monthly during the

ice-free months (May-December) and weekly during the first month to more accurately assess early changes in the litter. The initial plant material and litter bag contents were rinsed in distilled water and dried to a constant weight and ground in a Wiley mill in preparation for nutrient analyses. Samples were analyzed for total Kjeldahl nitrogen and total phosphorus and percent loss on ignition (LOI) was determined.

4.9 Nuisance Organisms

The marsh environment with its sheltered water surfaces provides a potentially ideal habitat for mosquitoes. To gather data on mosquito populations associated with marshland sewage treatment, a study was conducted at the Listowel artifical marsh facility in the summer of 1982 by Dr. G. A. Surgeoner of the University of Guelph. The objectives of the investigation were to determine 1) absolute numbers of mosquitoes produced in the marsh systems, 2) whether these mosquitoes constitute a nuisance or potential health hazard and 3) the feasibility of using Bacillus thuringiensis for control of mosquito larvae.

Mosquito numbers were determined from counts of larval and emerging insects in the channels of System 4, although observations were made in all systems. New Jersey light traps were set up on a continuous basis at varying distances from the marsh facility to investigate the extent of dispersal of the adult mosquito population in the vicinity of the marsh. The efficacy of the bacterial pesticide, B. thuringiensis var. isrealiensis was tested with dosages of .5-10 ppm applied to the marsh water. The potency of the dosages was evaluated by mortality of mosquito larvae placed in bioassay cages, as well as by sampling marsh mosquito larvae at 24-hour intervals after application. Detailed methods are described elsewhere (Surgeoner, 1982).

Muskrats were attracted to the Listowel marsh systems. Because their habit of tunneling into the berms has the potential for redirecting the path of the water flow, muskrat numbers were reduced by trapping.

5.0 RESULTS AND DISCUSSION

5.1 Operational Conditions

5.1.1 Hydraulic loadings and retention time

Wastewater renovation in marshes is accomplished in large part by the microbial-substrate complex. Previous studies indicate that in this environment, the mechanisms of wastewater treatment, which include sedimentation, precipitation, adsorption and microbial uptake and transformation, reduce contaminant levels as water circulates through a marsh. Plants, as part of the standing crop or in the form of litter, provide sites for microbial colonization, in addition to their role in nutrient uptake. Wastewater retention in the marsh must therefore be long enough for interaction between substances in the wastewater and the various marsh components.

The operation of the marsh systems permitted control of retention time by the regulation of water depth, in addition to the adjustment of hydraulic loadings. The Listowel marsh systems were operated with the objective of maintaining the wastewater residence time within the range of 7 to 10 days. Previous studies demonstrated high removal efficiencies within this range (Spangler et al, 1976; Nemgen, 1979).

At a constant hydraulic loading, theoretical retention times were altered by two major environmental factors, evapotranspiration and ice formation. High evapotranspiration rates in summer significantly increased the wastewater retention time causing stagnation and anoxia. This was particularly evident in the unusually dry summer of 1983 when oxygen stress was more severe than any other time during the four years. In Listowel, evapotranspiration during warm, dry weather often exceeded 60% of the daily hydraulic loading. In order to increase the rate of flow through the marshes, the water depth was lowered to 5-20 cm in

summer. Due to the presence of a lip on the outflow chambers, the water level was prevented from falling below a depth of 5 cm. If it had been possible to lower the water level to the level of the sediment, the effluent end of the marshes would probably have undergone intermittent drying during the summer period. The resultant increase in sediment oxygen content may have contributed to greater contaminant removal in summer.

Ice formation reduced the operational marsh volume resulting in a decreased detention time. To counteract the short-circuiting of wastewater due to ice blockage, water levels were raised to $\geq$ 30 cm prior to the onset of winter. The operational water level during the period of ice cover varied between winter averages of 17 and 26 cm, depending on the thickness of the ice. Heavy rains may have temporarily reduced retention time but the resulting dilution tends to mitigate losses in treatment efficiency.

The wastewater loadings averaged 86 m^3/d (1L/sec) in System 1 (shallow marsh, pond and channelized marsh complex) and 17 m^3/d (0.2 L/sec) in Systems 2-5 for most of the study period. The wastewater loadings were increased by 2-3 times at the start of winter in the first year to prevent freezing. However, the treatment efficiency was reduced by the higher loadings. In the following winters, aerators were installed in the culverts and propane heaters were placed in the effluent chambers to prevent interference of marsh operation by ice. The flow rates were returned to their original levels in the following summer. They were increased again slightly at the end of the four year period (summer, 1984) to adjust the retention time. Daily inflow and outflow discharge of wastewater in System 3 during the first two years of the study is illustrated in Figures 5.1 and 5.2.

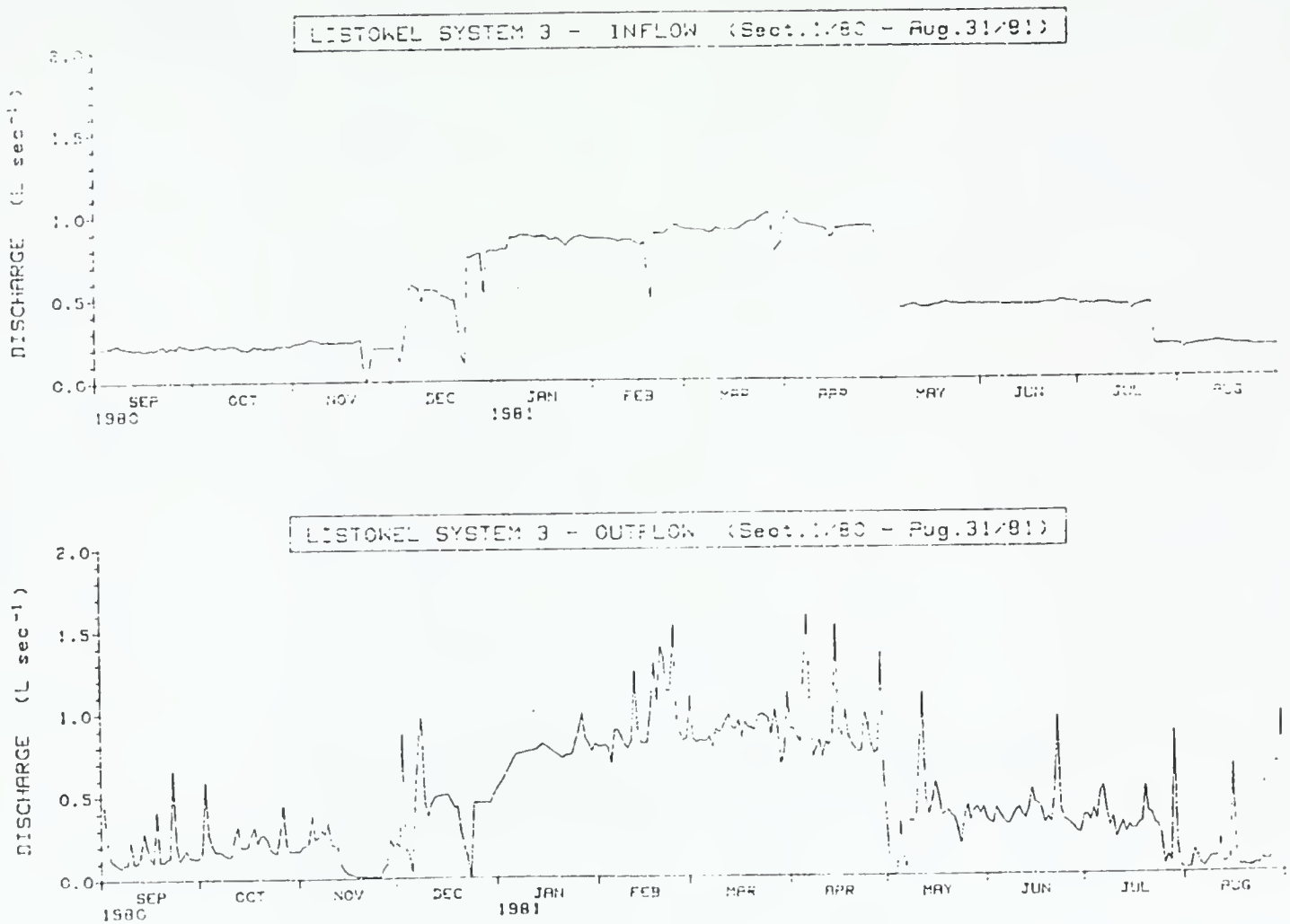


Figure 5.1 Daily inflow and outflow discharge in System 3 from September 1980 through August 1981.

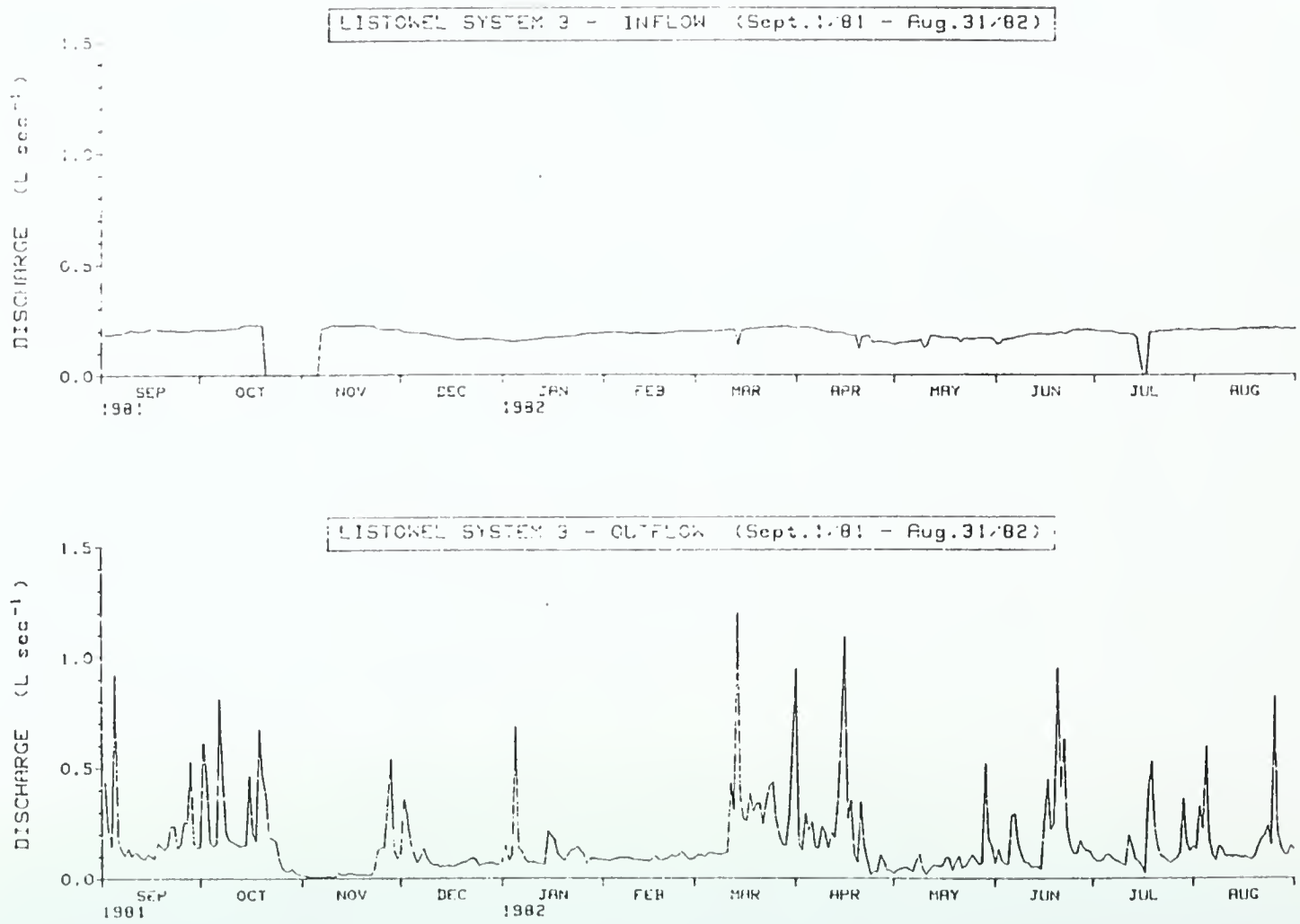


Figure 5.2 Daily inflow and outflow discharge in System 3 from September 1981 through August 1982

The theoretical residence time for each system was calculated from marsh volume and the average daily rate of inflow. The theoretical retention times in the marsh systems (ranging from 2 to 26 days) were often not realized due to environmental conditions, i.e. rainfall, evapotranspiration and ice formation. Based on the initial 14 cm water depth maintained in August through November, 1980, the theoretical residence time was 12.6 days in System 1, 7.1 and 7.4 days in System 2 and 5, respectively, and 10.4 days in the channelized marshes, Systems 3 and 4.

Changes in operational conditions, i.e. flow rates, water depth and retention time, over the four years are listed in Table 5.1. For this purpose, the four years were subdivided into periods with similar operational conditions which do not coincide with seasonal definitions used elsewhere in this report. Figure 5.3 demonstrates the impact of flow and water depth on the theoretical retention time.

Dye studies provided a measure of actual residence time. Although these studies were not conducted on a regular basis, dye study estimates of retention time ranged from 3.3 to 15.9 days (Table 5.1). Actual retention was generally longer than the theoretical retention time in summer (due to the effect of evapotranspiration on flow reduction) and shorter in winter due to underestimates of marsh volume reduction from ice formation. For this reason, theoretical retention estimates of 13-15 days in the summer of 1983 probably were associated with actual retention times exceeding 15 days.

Table 5.1 Operational Conditions in the Marsh Systems

System 1

Year	Period	Average Inflow Rate (m^3/d)	(m^3/ha/d)	Water Depth* in Cells#1&#3 (cm)	Theoretical** Retention(d)	Actual*** Retention(d)
1980	Aug. 19-Nov. 15	86	206	14	12.6	-
1980/81	Nov. 15-May 14	200	479	30	8.0	-
	Period of ice cover			13	3.7	-
1981	May 14-Nov. 5	176	422	20	7.3	
1981/82	Nov. 5-Apr. 14	86	206	34	20.2	-
	Period of ice cover			(20-31)	(14.4-18.7)	-
1982	Apr. 14-June 16	86	206	20	14.9	-
1982	June 16-Nov. 2	86	206	5	9.1	-
1982/83	Nov. 2-Mar. 16	86	206	33	19.8	-
	Period of ice cover			(20-29)	(14.5-18.1)	-
1983	Mar. 16-Nov. 2	86	206	20	14.9	-
1983/84	Nov. 2-Apr. 6	86	206	32	19.4	-
	Period of ice cover			(18-29)	(14.2-17.7)	-
1984	Apr. 6-Aug. 7	86	206	16	13.3	-

System 2

Year	Period	Average Inflow Rate (m^3/d)	(m^3/ha/d)	Water Depth* (cm)	Theoretical** Retention(d)	Actual*** Retention(d)
1980	Aug. 19-Nov. 15	17	187	14	7.1	-
1980/81	Nov. 15-May 14	68	748	30	3.8	-
	Period of ice cover			15	1.9	-
1981	May 14-July 31	32	352	20	5.4	5.4(June 8)
1981	July 31- Nov. 5	17	187	20	10.2	8.8(July 31)
1981/82	Nov. 5-Apr. 14	17	187	34	17.3	-
	Period of ice cover			27(19-35)	13.7(9.7-17.8)	5.4(Jan. 29)
1982	Apr. 14-Nov. 2	17	187	20	10.2	10.7(Sept. 7)
1982/83	Nov. 2-Mar. 16	17	187	34	17.3	-
	Period of ice cover			26(25-30)	13.2(12.7-15.2)	-
1983	Mar. 16-Jun 30	17	187	20	10.2	10.0(May 26)
1983	June 30-Nov. 2	17	187	10	5.1	11.7(Sept. 7)
1983/84	Nov. 2-Apr. 6	17	187	35	17.8	-
	Period of ice cover			17(15-22)	8.6(7.6-11.2)	-
1984	Apr. 6-Aug. 7	22	242	10	3.9	-

Table 5.1 Operational Conditions in the Marsh Systems (Cont'd)

System 3

Year	Period	Average Inflow Rate (m^3/d)	(m^3/ha/d)	Water Depth* (cm)	Theoretical** Retention(d)	Actual*** Retention(d)
1980	Aug. 19-Nov. 15	17	128	14	10.4	-
1980/81	Nov. 15-May 14	70	529	30	5.4	-
	Period of ice cover			18	3.2	-
1981	May 14-Nov. 5	38	287	20	6.6	
1981/82	Nov. 5-Apr. 14	17	128	34	25.2	-
	Period of ice cover			22(16-32)	16.3(11.8-23.7)	13.4(Feb. 5)
1982	Apr. 14-June 16	17	128	20	14.8	-
1982	June 16-Nov. 2	17	128	5	3.7	14.0 (Oct. 12)
1982/83	Nov. 2-Apr. 5	17	128	34	25.2	-
	Period of ice cover			26(24-29)	19.2(17.8-21.5)	-
1983	Apr. 5-July 26	17	128	18	13.3	-
1983	July 26-Nov. 2	17	128	15	11.1	10.4(Aug. 5)
1983/84	Nov. 2-Apr. 6	17	128	33	24.4	-
	Period of ice cover			20(14-24)	14.8(10.4-17.8)	15.9(Jan. 4)
1984	Apr. 6-July 19	21	159	12	7.2	12.8(May 31)
1984	July 19-Aug. 7	21	159	17	10.2	>15.0(July 13)

System 4

Year	Period	Average Inflow Rate (m^3/d)	(m^3/ha/d)	Water Depth* (cm)	Theoretical** Retention(d)	Actual*** Retention(d)
1980	Aug. 19-Nov. 13	17	128	14	10.4	-
1980/81	Nov. 13-May 14	50	378	30	7.6	-
	Period of ice cover			17	4.3	-
1981	May 14-Nov 5	35	264	20	7.2	6.3(June 25)
1981/82	Nov. 5-Apr. 14	17	128	34	25.2	-
	Period of ice cover			23(19-27)	17.0(14.1-20.0)	12.2(Jan. 7)
1982	Apr. 14-June 16	17	128	20	14.8	-
1982	June 16-Nov. 2	17	128	5	3.7	9.4(Sept. 28)
1982/83	Nov. 2-Mar. 15	17	128	35	25.9	-
	Period of ice cover			24(23-27)	17.8(17.0-20.0)	7.8 (Jan. 7)
1982	Mar. 15-July 4	17	128	20	14.8	-
1983	July 4-Nov. 2	26	196	20	9.7	8.8 (Aug. 5)
1983/84	Nov. 2-Apr. 6	26	196	32	15.5	-
	Period of ice cover			20(17-22)	9.7(8.2-10.6)	13.1(Jan. 4)
1984	Apr. 5-May 1	26	196	11	5.3	-
1984	May 1-July 19	33	249	15	5.7	13.6 (June 7)
1984	July 19-Aug. 7	34	257	14	5.2	13.0 (Aug. 13)

Table 5.1 Operational Conditions in the Marsh Systems (Cont'd)

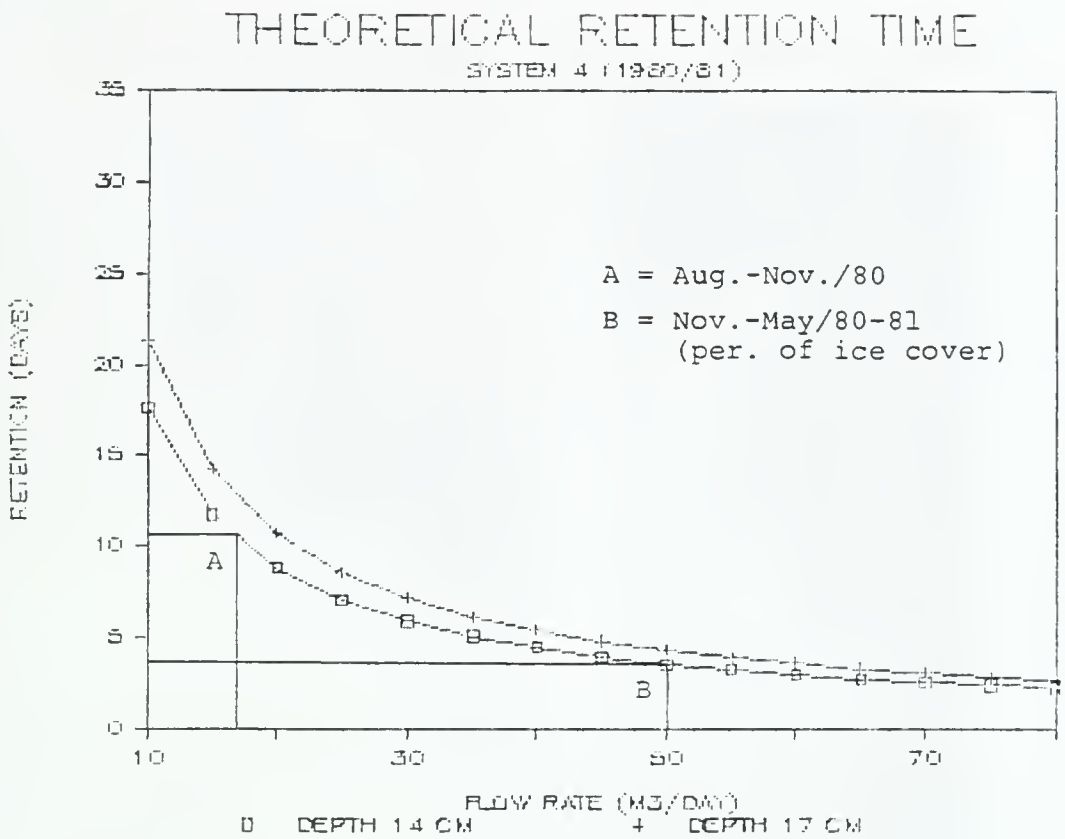
System 5

Year	Period	Average Inflow Rate (m^3/d)	(m^3/ha/d)	Water Depth* (cm)	Theoretical** Retention(d)	Actual*** Retention(d)
1980	Aug. 19-Nov. 13	17	181	14	7.4	-
1980/81	Nov. 13-May 14	53	563	30	5.1	-
	Period of ice cover			11	1.9	-
1981	May 14-July 31	30	319	20	6.0	3.8 (June 8) 4.3 (July 8)
1981	July 31-Nov. 5	17	181	20	10.5	13.0 (July 31)
1981/82	Nov. 5-Apr. 14	17	181	34	17.9	-
	Period of ice cover			26(20-31)	13.7(10.5-16.3)	5.6(Jan.21)
1982	Apr. 14-May 14	17	181	20	10.5	-
1982	May 14-July 16			Shut down		
1982	July 16-Nov. 2	17	181	5	2.6	3.3 (Nov. 1)
1982/83	Nov. 2-Mar. 16	17	181	34	17.9	-
	Period of ice cover			27(25-29)	14.2(13.1-15.2)	8.2 (Jan. 7)
1983	Mar. 16-July 4	17	181	20	10.5	9.2 (May 27)
1983	July 4-Nov. 2	26	276	20	6.9	6.7(Sept. 7)
1983/84	Nov. 2-Apr. 6	26	276	34	11.7	-
	Period of ice cover			19(13-26)	6.5(4.5-8.9)	-
1984	Apr. 6-May 1	27	287	6	2.0	-
1984	May 1-Aug. 7	34	361	6	1.6	-

* Water depths, in most cases, represent settings rather than actual measurements for ice-free periods. Deep pond in System 1 was set at 90 cm. Shallow marsh in System 1 assumed to be at same depth as channelled marsh. Range of water depths during period of ice cover in brackets.

** Calculation allowed for 5% volume reduction due to cattail growth and litter. Volume alterations due to evapotranspiration and rainfall not included.

*** Retention based on dye studies.



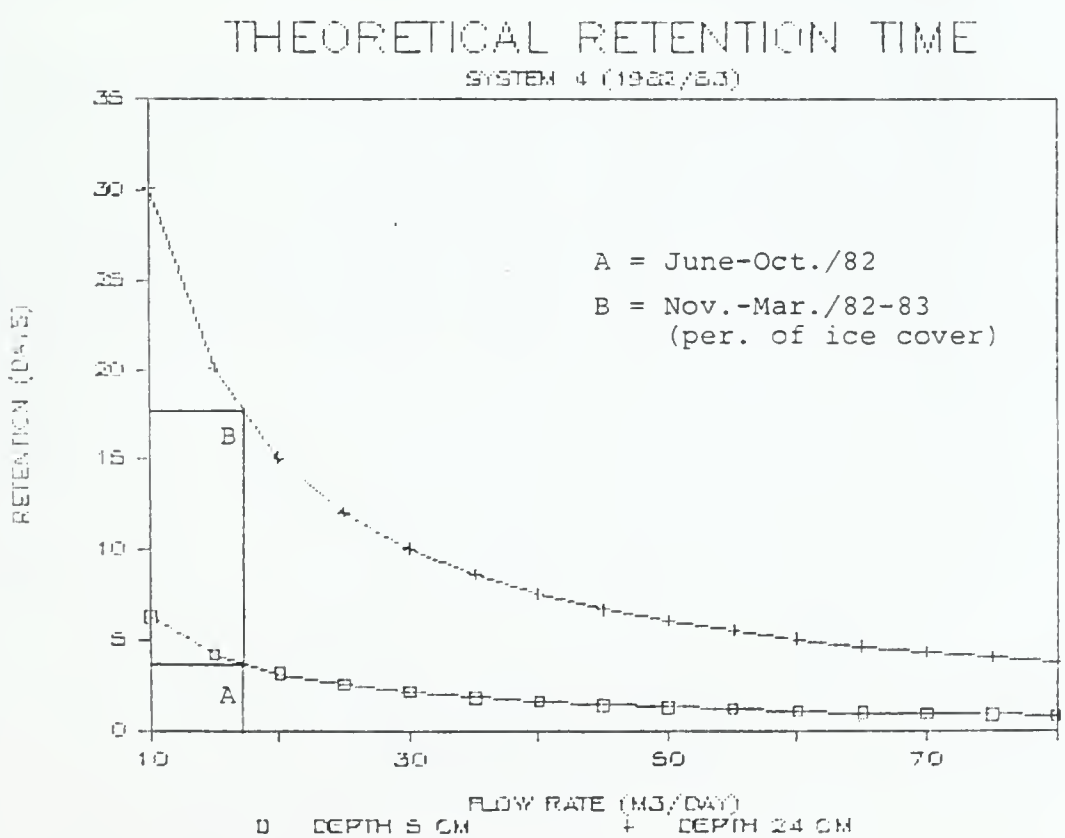


Figure 5.3 Theoretical hydraulic retention as a function of flow rate and water depth in System 4.

Considerable short-circuiting was occurring in Systems 2 & 5 (shallow marshes) as indicated by the short retention times (<7 days) measured in these systems. The construction of baffles in System 2 in the spring of 1982 substantially increased retention in the marsh. Whereas 5 day retentions were measured in summer and winter prior to construction, subsequently dye studies indicated retention times between 10 and 11 days. The short-circuiting that occurred prior to the installation of baffles is demonstrated in diagrams derived from a sampling grid of relative dye intensity following dye injection in System 2 (Figure 5.4). At the time of the dye study, hydraulic detention time was 7 days and wastewater loadings in System 2 were 180 m^3/ha/d.

5.1.2 Hydrologic budget

Seasonable and annual water balances for all systems are summarized in Table 5.2. The percent difference between the total hydrologic input and output in each marsh system was based on measurement of inflow and outflow volumes plus gains (i.e., precipitation and drainage) and losses (i.e., evapotranspiration) due to environmental conditions.

During the first year of marsh operation when wastewater loadings were high, the annual hydrologic balance ranged between +4% (i.e., 4% net gain) and - 10% (i.e., 10% net loss). In the following years the percent difference between total hydrologic input and output indicated that more than 10% of the water entering Systems 3, 4 and 5 was unaccounted for. The percent difference remained low in Systems 1 and 2 during all 4 years (-1 to -7%).

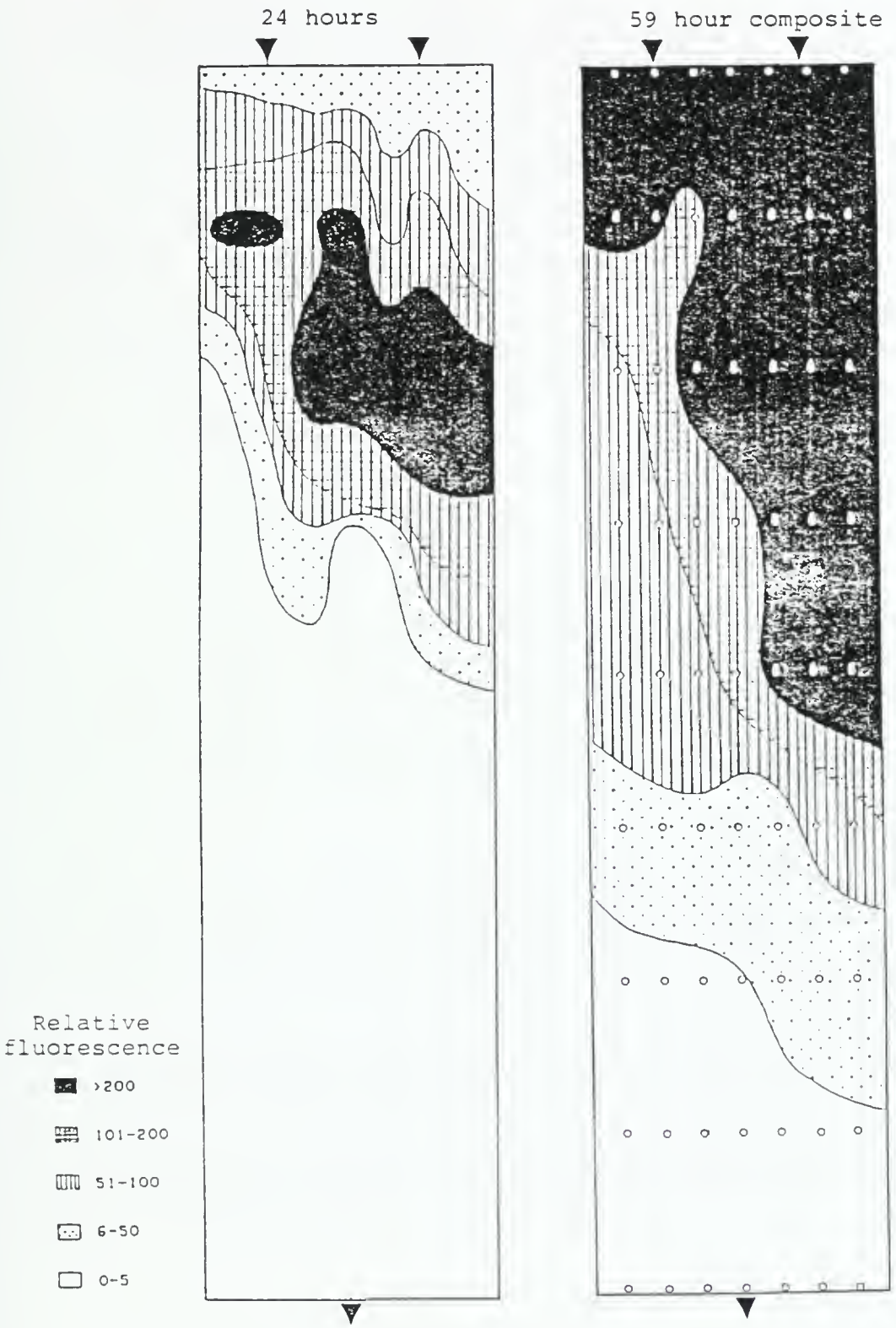


Figure 5.4 Relative dye intensities in System 2 after 24 hours and composite of maximum intensities of samples collected up to 59 hours in May, 1982. (Circles indicate sampling sites)

Table 5.2 Percent Difference Between Total Hydrologic Input and Output in the Marsh Systems*

	Systems				
Year/Season	1	2	3	4	5
1980-81					
Fall	-6	-13	-27	-21	-14
Winter	-4	+3	-7	+3	+0.2
Spring	-1	+18	-3	+8	-0.7
Summer	-0.4	-10	-16	-29	-23
Annual	-3	+4	-10	-5	-6
1981-82					
Fall	-2	-16	-23	-26	-15
Winter**	-15	-18	-56	-47	-29
Spring	+7	+15	-6	-1	+17
Summer	-4	-7	-16	-39	-27
Annual	-4	-7	-25	-29	-14
1982-83					
Fall	-4	-17	-33	-38	-40
Winter	-2	-9	-34	-19	-12
Spring	+1	-5	-20	-18	-15
Summer	-13	-6	-31	-35	-20
Annual	-4	-9	-29	-27	-22
1983-84					
Fall	-5	-19	-34	-13	-10
Winter	-3	-15	-39	-8	+1
Spring	+8	+8	-14	-1	+9
Summer	-3	-1	-22	-29	-7
Annual	-1	-6	-27	-13	-2

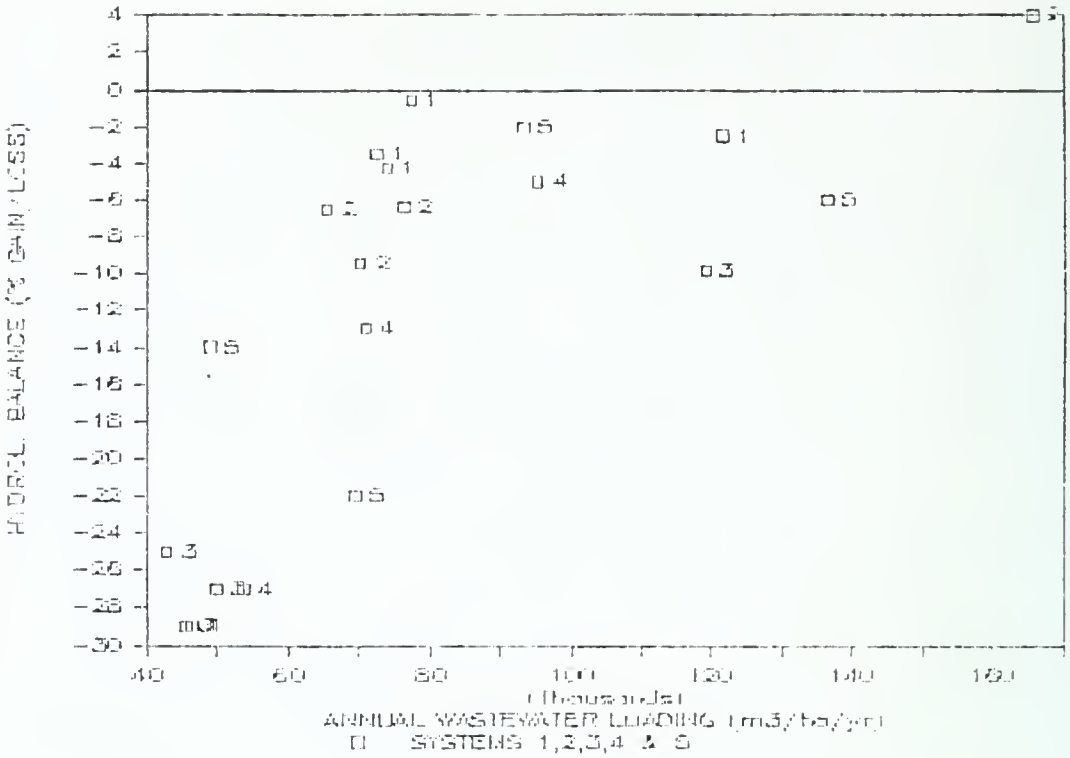
* Percentages calculated from hydrologic measurements and estimates as follows: (Input-Output)/Input. Minus sign indicates proportion of input unaccounted for.

** The accuracy of data during this season is questionable as frost heaving was observed on all flow recording equipment.

The difference between total hydrologic input and output was greater in systems with lower hydraulic loadings. Systems showing the highest annual losses (>25%), System 3 (in 3 years) and System 4 (in 2 years), were clustered at the lower end of the scale of annual outflow discharge and annual wastewater loading (Figure 5.5). Wastewater loadings were highest in Systems 1 and 2, followed by System 5, and were lowest in Systems 3 and 4. For most of the study, loadings for Systems 1 through 5 were 206, 187, 128, 128 and 180 m^3/ha/d, respectively. Following the first year, monthly wastewater inflow volumes in each system were relatively constant. As in the example of System 3 (Figure 5.6), the monthly outflow discharges for the second and third years were inversely related to calculated net losses in the hydrologic budget. Monthly inflow rates were relatively constant.

These data suggest errors in the system outflow measurements. Low hydraulic loadings and outflow discharge rates were associated with negative hydrologic balances as indicated in the comparison between systems or monthly data within systems. For this reason the outflow discharges used in calculating effluent loadings were adjusted, i.e., they were multiplied by the ratio of total hydrologic input over output.

Because of the reduced wastewater loadings in years 2 through 4, a higher proportion of the hydrologic budget was represented by precipitation, drainage and evapotranspiration during this period. The wastewater influent provided the major hydraulic input to the marsh systems, with precipitation ranging between 5 and 6% of the total annual input as measured in the first year and 9-18% in the following years. The contribution of terrestrial runoff was calculated as 1-3% of the total hydraulic loading in the first year and 2-8% in other years, assuming that 40% of the precipitation in the surrounding area drained into the marshes.



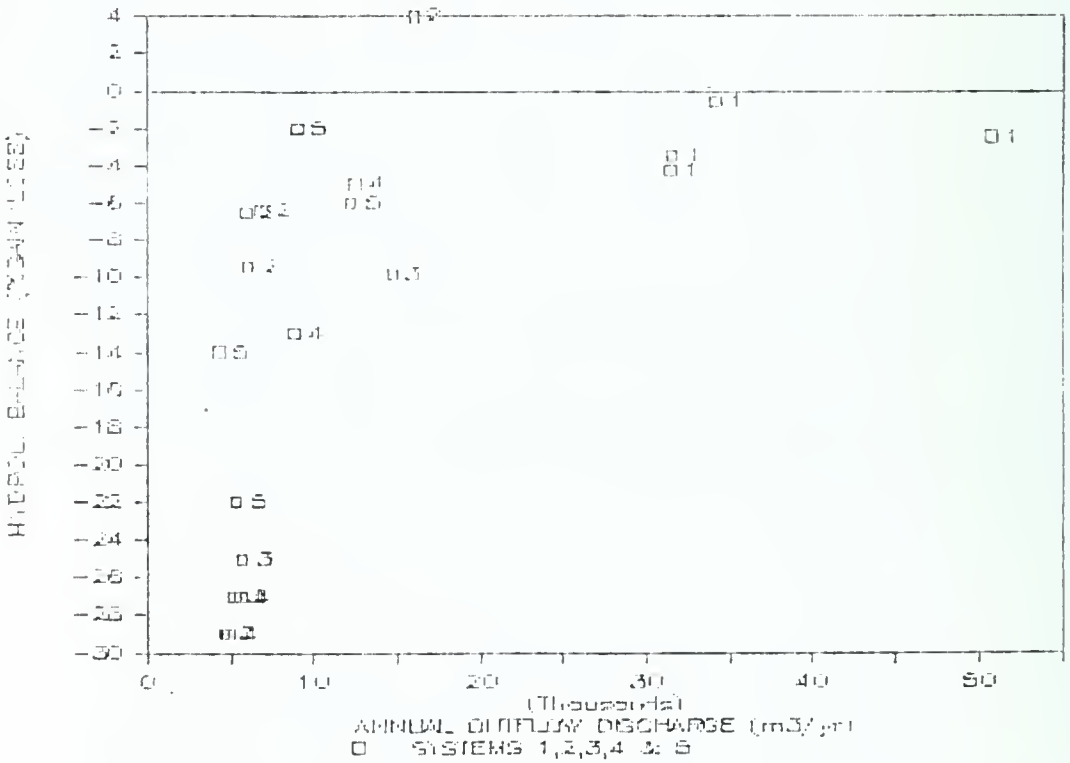
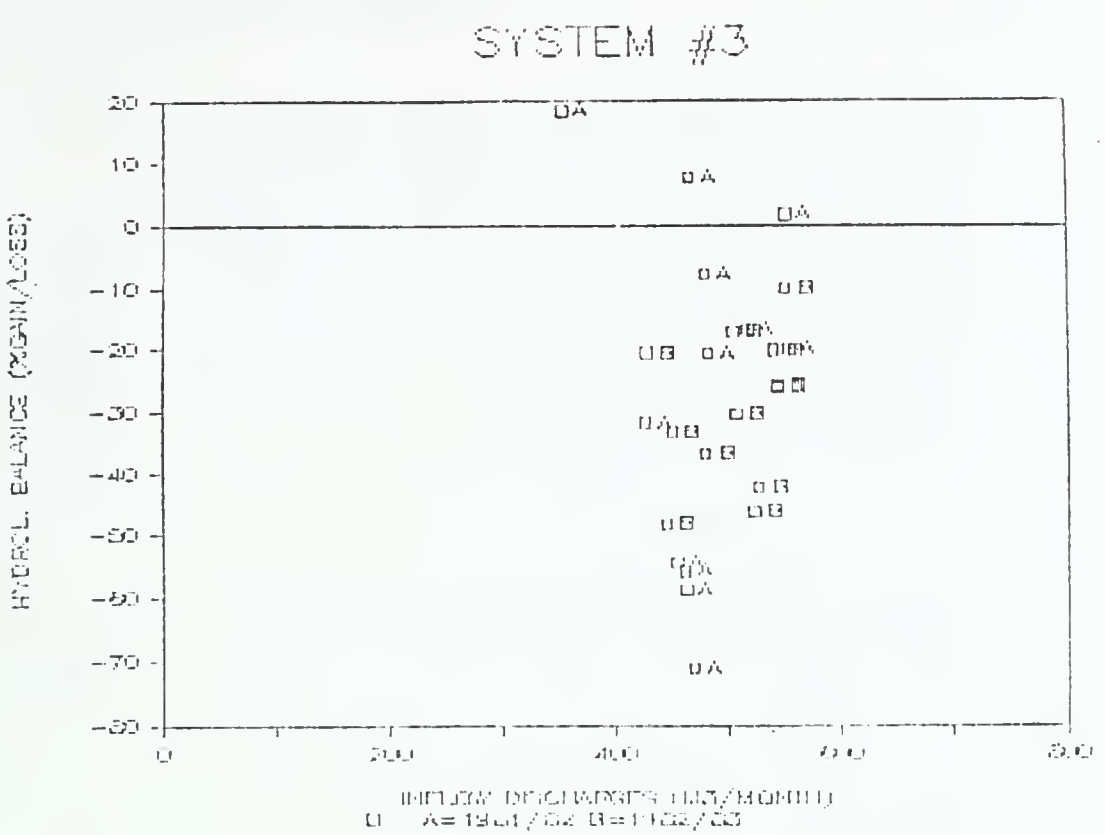


Figure 5.5 Hydrologic balance vs. annual outflow discharge rate and annual wastewater loading in marsh systems during 4 years.



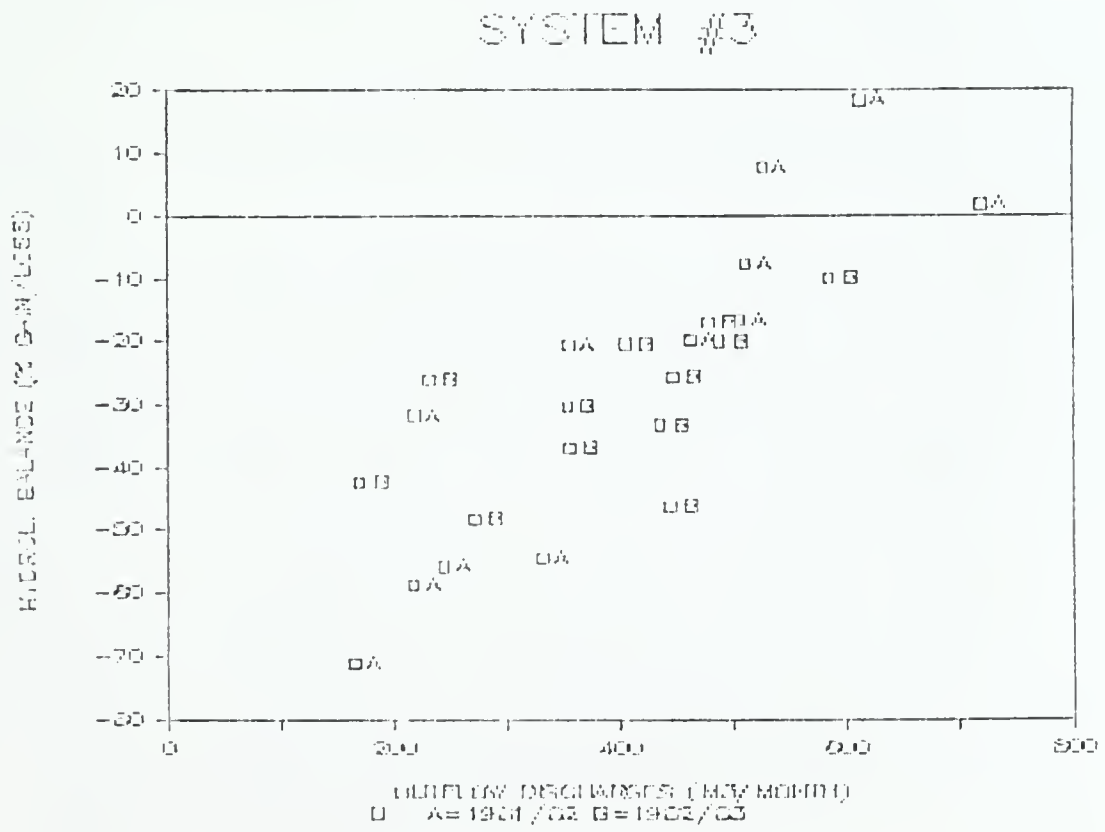


Figure 5.6 Hydrologic balance vs. monthly inflow and outflow discharge rates in System 3.

Marsh effluent flowing through the outflow weirs represented the major removal of water from the systems. Estimates of evapotranspiration accounted for 4-7% of the total annual water volume removed from the marshes in the first year and 8-15% in the succeeding years. The percent of the total annual hydrologic input lost through evapotranspiration in System 3 was 5% in the first year and 11-12% in the following years. The loss of water in harvested plants was trivial (.02% or less).

5.1.2 Influent water quality

Two types of wastewater pre-treatment were tested in the Listowel marsh systems to evaluate marsh treatment capabilities. Influent characteristics during the 4 year period are summarized in Table 5.3.

The raw sewage received continuous alum injection prior to 3.5 days in the mixed aeration cell. The alum addition resulted in precipitation of soluble phosphates into water insoluble forms, predominantly metal phosphates, but the constant mixing kept total phosphorus and suspended sediment levels high in the aeration cell effluent. Sedimentation of suspended particles occurred after entry into Systems 4 and 5. The 3.5 day mixed aeration reduced BOD_5 levels to some extent but did not support nitrification or algae growth.

The lagoon effluent contained lower levels of most wastewater constituents. The suspended solids and BOD_5 were one-fifth and one-third, respectively, of levels in the aeration cell effluent. Maximum oxygen concentrations in the lagnon effluent were approximately three times (up to 23 mg/L) greater than maximum concentrations in the aeration cell effluent due to the supersaturation of oxygen from algae in the lagoon. Annual average oxygen concentrations ranged between 2.9 and 4.5 times higher in the lagoon effluent. Average ammonia levels during the four year perind were only slightly lower in the lagoon effluent then in the aeration cell effluent. Hydrogen sulfide levels were considerably higher in the lagoon effluent, particularly in the winter months.

Table 5.3 Lagoon Effluent and Aeration Cell Effluent Quality

WASTEWATER CHARACTERISTICS	AERATION CELL EFFLUENT MEAN (4 yr)	AERATION CELL EFFLUENT RANGE*	LAGOON EFFLUENT MEAN (4 yr)	LAGOON EFFLUENT RANGE*	MEAN AERATION CELL MEAN LAGOON EFFLUENT
TEMPERATURE (^{o}C)	12.45	3.0 - 26.0	10.4	0. - 27.0	1.2
PH	7.3	6.3 - 8.8	7.5	5.7 - 9.1	1.0
CONDUCTIVITY (umhos)	1051.5	550. -1600.	985.4	550. -1450.	1.1
SUSPENDED SOLIDS (mg/L)	111.1	14.5 - 406.7	22.8	0.8 - 98.8	4.9
BOD_5 (mg/L)	56.3	10.1 - 168.0	19.6	2.6 - 69.0	2.9
DO (mg/L)	2.2	0. - 7.8	5.7	0. - 23.3	0.4
TP (mg/L)	3.2	0.8 - 9.1	1.0	0.3 - 2.4	3.2
SRP (mg/L)	0.40	0.01- 3.0	0.37	0.005- 1.7	1.1
TKN (mg/L)	18.7	7.0 - 37.0	12.0	5.0 - 26.5	1.6
TOTAL NH_3 (mg/L)	8.6	1.2 - 19.7	7.2	0.1 - 16.5	1.2
NO_2+NO_3 (mg/L)	0.38	<0.01- 16.2	0.25	<0.01 - 7.2	1.5
HYDROGEN SULFIDE (mg/L)	0.23	<0.01- 6.2	1.81	<0.01 - 10.8	0.1
PHENOLS (ug/L)	3.3	<1.0 - 11.5	4.2	<1.0 - 16.5	0.8

* Derived from weekly and biweekly samples.

5.1.4 Temperature, pH and chlorophyll a levels

Monthly average temperatures in the influent and effluent of Systems 3 and 4 during the four years are indicated in Figures 5.7 and 5.8. Temperatures were similar in each year for all systems receiving the same type of influent.

Summer temperatures in both the aeration cell effluent and lagoon effluent were similar and declined slightly in the experimental marshes. Temperatures in the marshes were highest in July; maximum July average was 19.6^{o}C. Winter temperatures in the aeration cell effluent were several degrees higher than the lagoon effluent but monthly average temperatures were similar in the effluent of all marsh systems in the winter months. In December through March temperatures in the marsh effluent were as low as 0^{o}C, but were in all cases less than 2^{o}C, except in February of 1983 when temperatures rose in response to elevated air temperatures.

Monthly average pH values measured in Systems 3 and 4 are illustrated in Figures 5.9 and 5.10. The lagoon effluent pH ranged between 6.2 and 8.5 and showed seasonal variations with peaks (mostly >8.0) in the late spring/early summer and fall. The elevated pH in the lagoon effluent resulted from the presence of algae.

In the process of photosynthesis, there is a net increase in dissolved oxygen in the water whereas carbon dioxide is removed from the water as it is utilized by algae as the carbon source for the production of carbohydrates. Monthly average chlorophyll a concentrations in Systems 3 and 4 are illustrated in Figures 5.11 and 5.12, respectively. System 4 influent levels were not tested but were close to zero as the result of the short detention and constant agitation in the aeration cell.

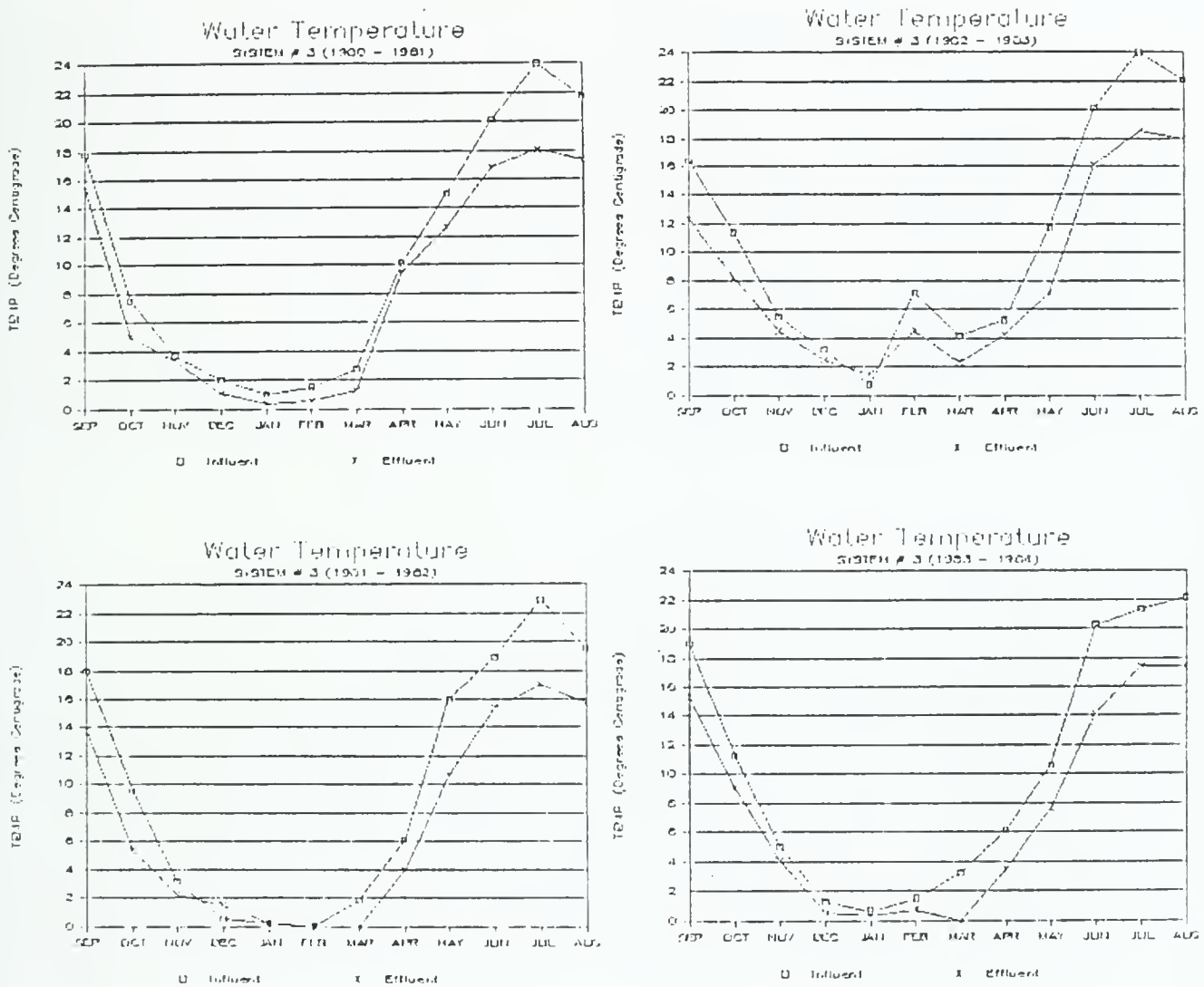

Water Temperature
SYSTEM # 3 (1980 - 1981)
Water Temperature
SYSTEM # 3 (1982 - 1983)
Water Temperature
SYSTEM # 3 (1981 - 1982)
Water Temperature
SYSTEM # 3 (1983 - 1984)
TEMP (Degrees Centigrade)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
I Influent
X Effluent

Figure 5.7 Monthly average water temperature of System 3 influent and effluent (4 years).

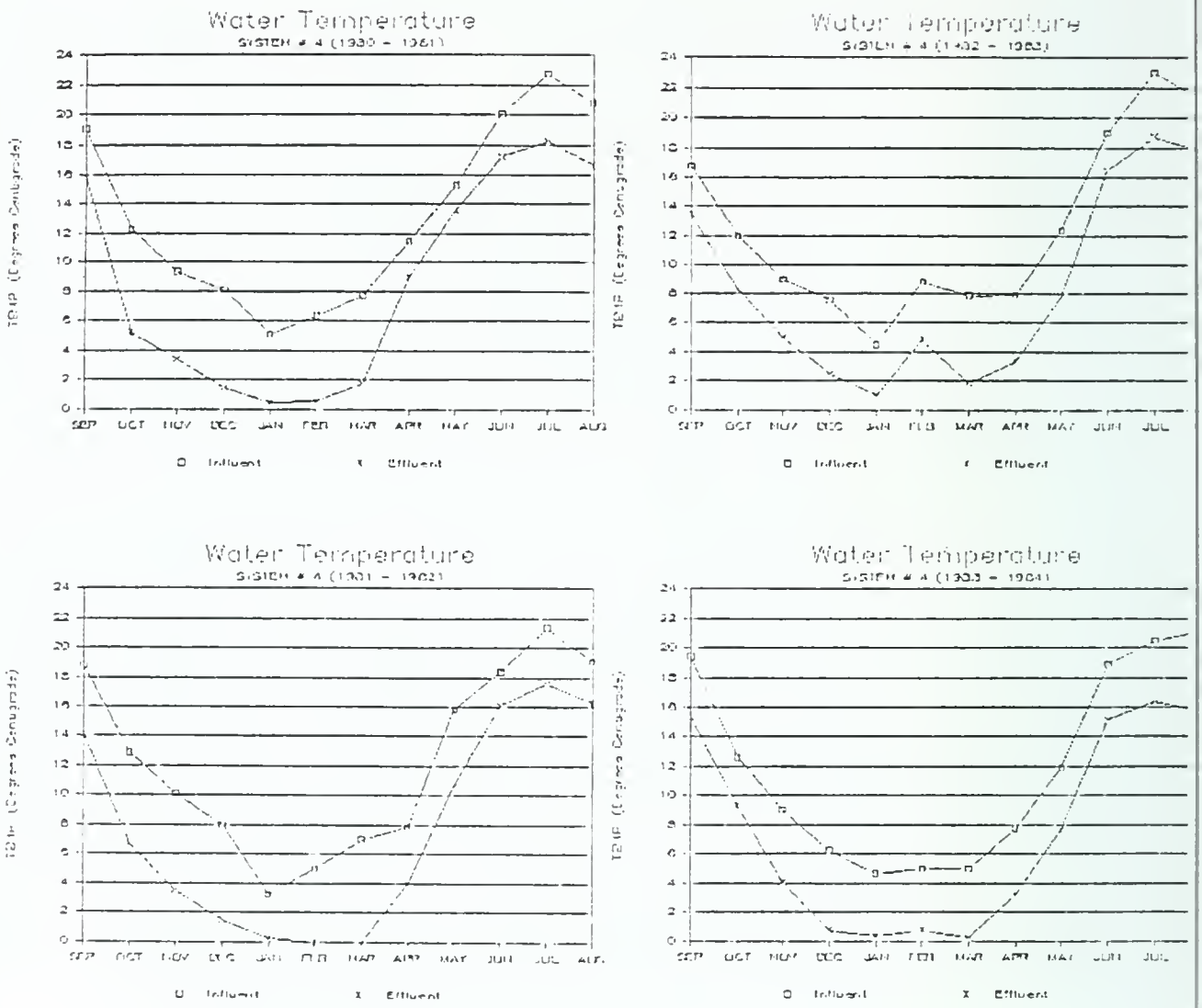

Water Temperature
SYSTEM # 4 (1980 - 1981)
Water Temperature
SYSTEM # 4 (1982 - 1983)
Water Temperature
SYSTEM # 4 (1981 - 1982)
Water Temperature
SYSTEM # 4 (1983 - 1984)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent

Figure 5.8 Monthly average water temperature of System 4 influent and effluent (4 years).

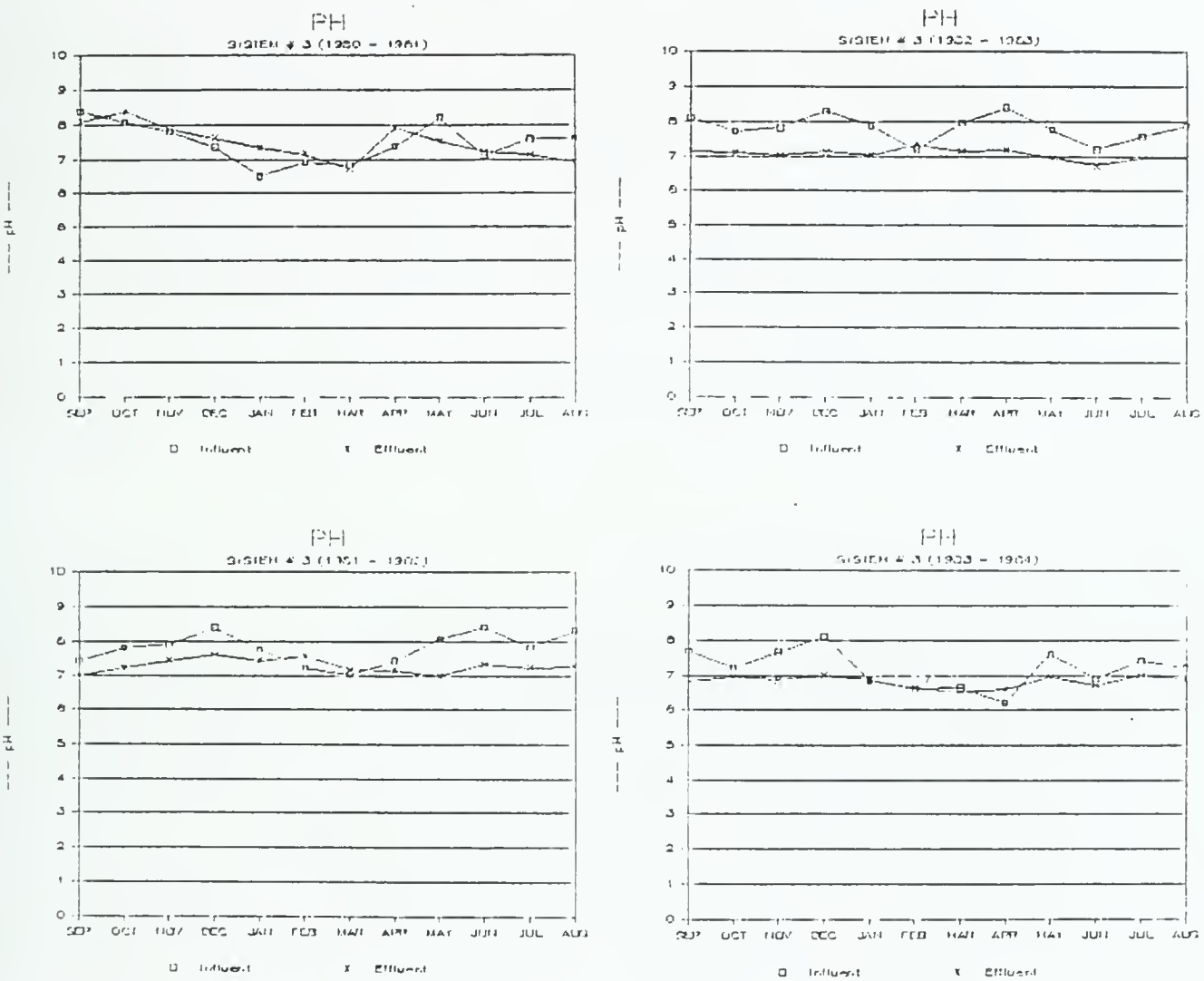


Figure 5.9 Monthly average pH of System 3 influent and effluent (4 years).

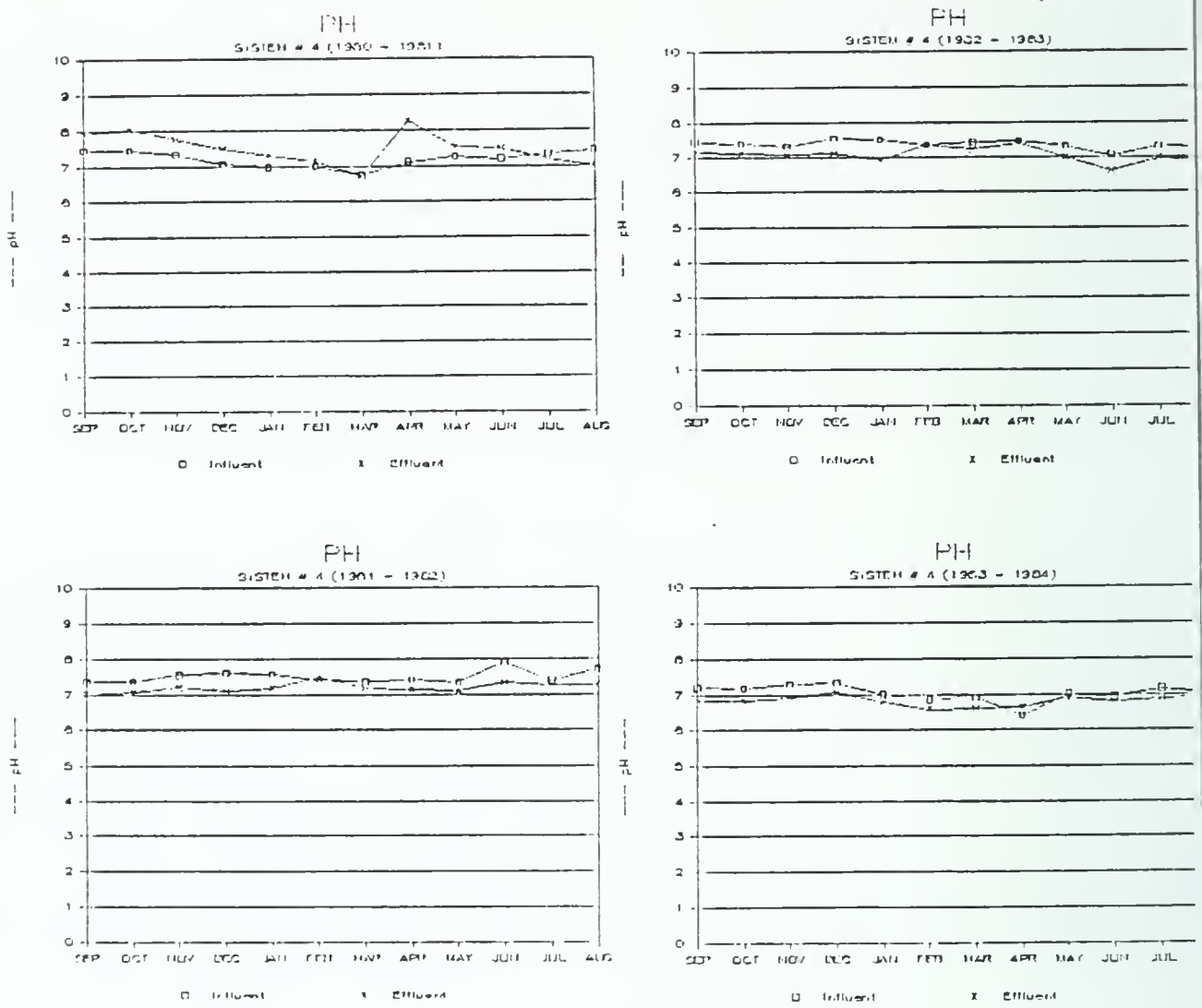

PH
SYSTEM # 4 (1960 - 1961)
PH
SYSTEM # 4 (1962 - 1963)
PH
SYSTEM # 4 (1961 - 1962)
PH
SYSTEM # 4 (1963 - 1964)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

Figure 5.10 Monthly average pH of System 4 influent and effluent (4 years).

The effluent pH of Systems 1-3 reflected the influent variation in the first year showing peaks in spring/summer and fall. The seasonal fluctuations were dampened in the succeeding three years so that effluent pH in these systems were generally lower than influent pH and showed little seasonal variation. Little or no seasonal pattern was apparent in the narrower annual pH range detected in the aeration cell effluent fed to Systems 4 and 5. PH values in the effluents of Systems 4 and 5 were similar to those in Systems 1-3, generally falling within the range of pH 6.5 - 7.5.

Wastewater temperature and pH affect the nature and rate of chemical and microbiological processes that provide contaminant removal in marshes. The specific impact of these parameters are described in the following chapters dealing with particular substances of importance in evaluating wastewater treatment.

5.2 Water Quality

5.2.1 Suspended Solids

Monthly average suspended solids concentrations and seasonal loadings are illustrated in Figures A.1 and A.2, respectively. Suspended solids concentrations in the aeration cell effluent were extremely high, samples ranging from 14 to 406 mg/L. Reduced levels were observed in one or more winter months in all years. The high suspended solids loadings to Systems 4 and 5 resulted in the formation of sludge mats, accompanied by some cattail die-back, near the influent ends of these systems. Pre-treatment, which allows settling of suspended solids prior to discharge in the marsh, would resolve this problem, as well as the high BOD_5 levels associated with the particulate organic matter in the suspended solids. Solids settling in Systems 4 and 5 was aided by coagulation due to alum additions prior to entry in the aeration cell.

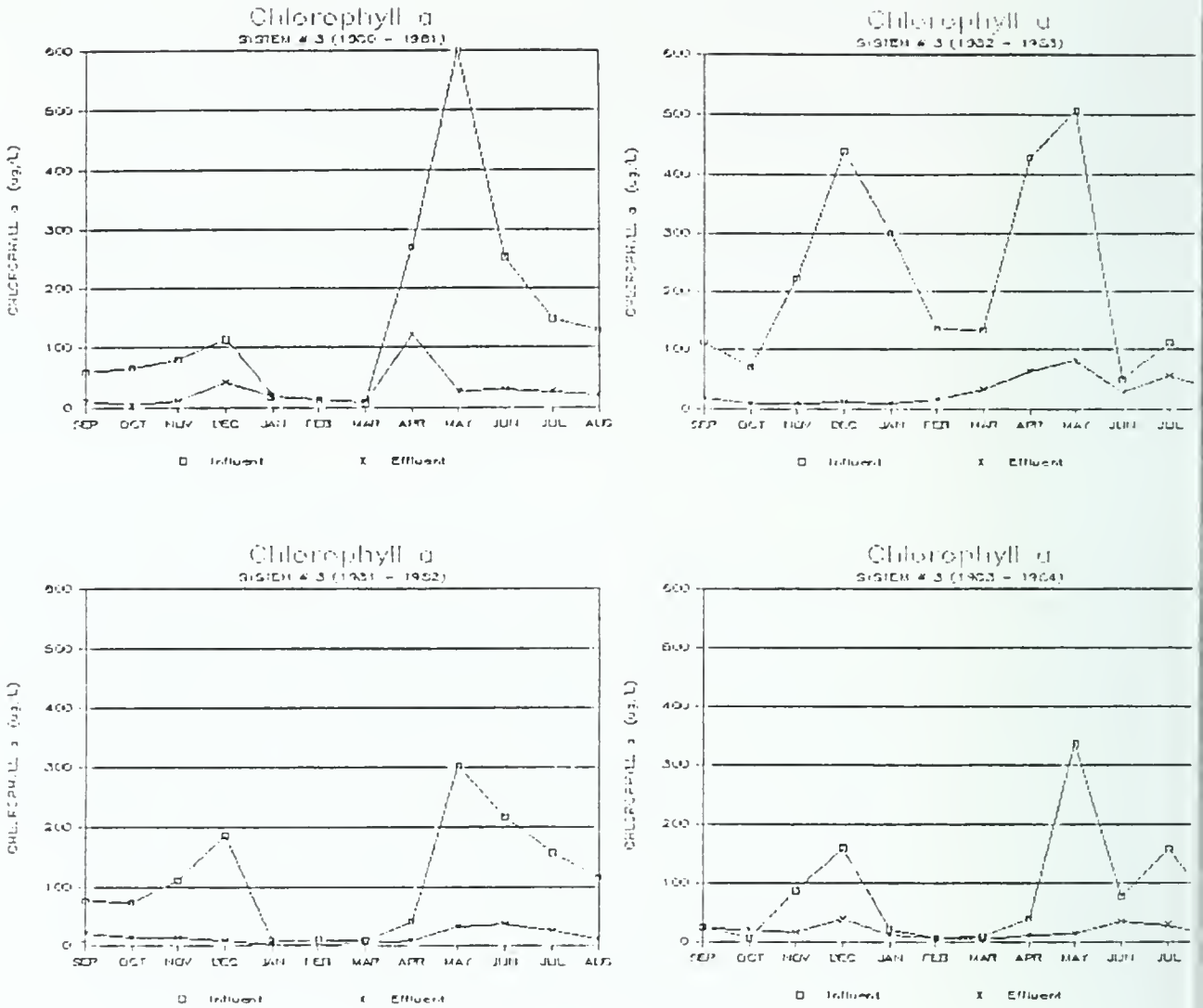

Chlorophyll a
SYSTEM # 3 (1980 - 1981)
Chlorophyll a
SYSTEM # 3 (1982 - 1983)
Chlorophyll a
SYSTEM # 3 (1983 - 1984)
CHLOROPHYLL a (ug/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

Figure 5.11 Monthly average chlorophyll a concentration in System 3 (4 years).

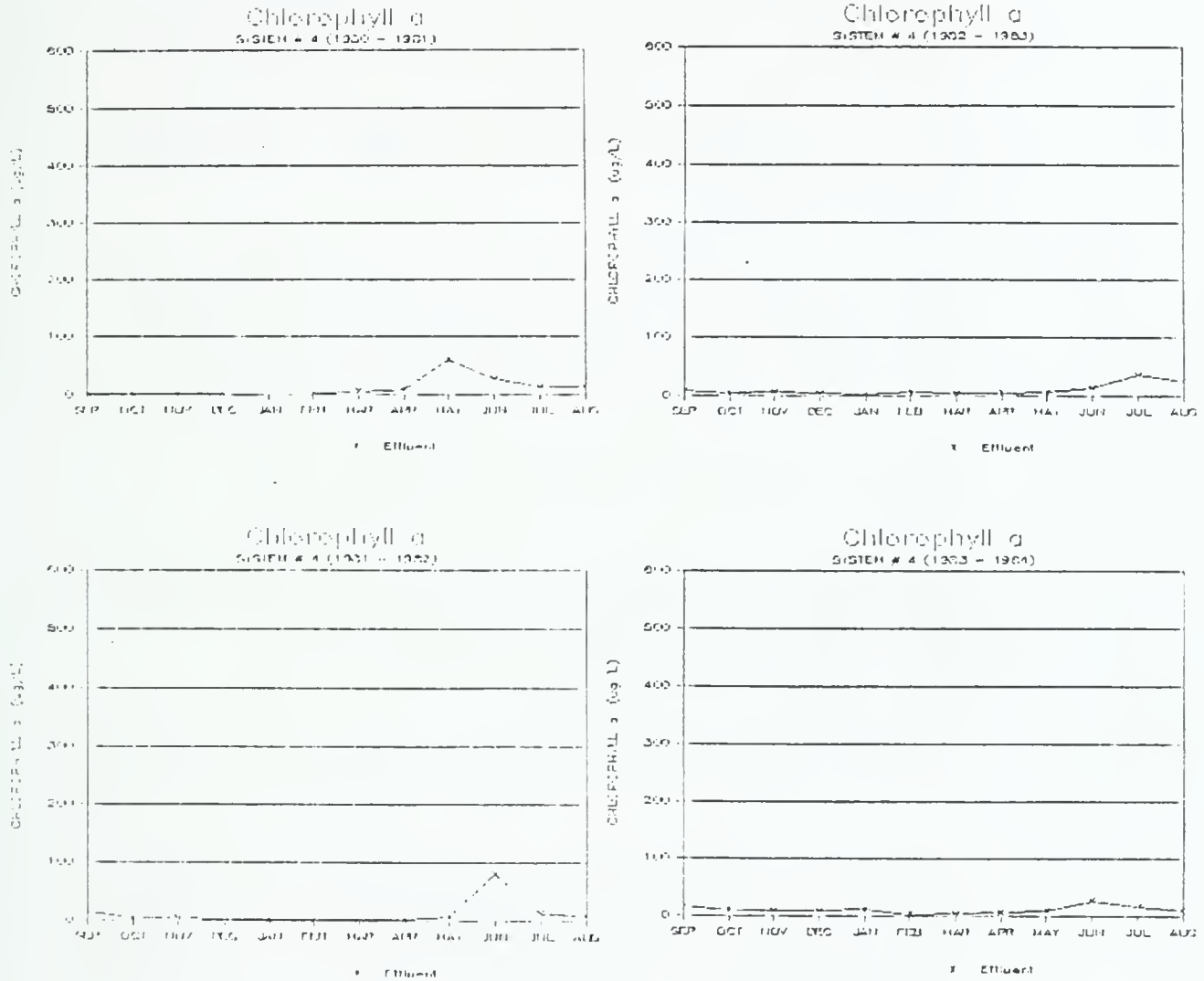

Chlorophyll a
SYSTEM # 4 (1980 - 1981)
Chlorophyll a
SYSTEM # 4 (1982 - 1983)
Chlorophyll a
SYSTEM # 4 (1983 - 1984)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Effluent

Figure 5.12 Monthly average chlorophyll a concentration in System 4 (4 years).

The lagoon effluent which was fed to Systems 1, 2 and 3 contained suspended solids concentrations generally less than 40 mg/L (samples ranging from 1 to 99 mg/L) due to previous settling in the lagoon. Sludge mats did not develop in the systems receiving lagoon effluent. Elevated loadings in spring and summer coincided with algae blooms in the facultative lagoons which at these times contained chlorophyll concentration peaks of 300-600 ug/L. Increases in lagoon effluent suspended solids levels observed in November through January were also coincidental with increases in chlorophyll levels.

Suspended solids concentrations in the marsh system effluents were generally below 15 mg/L from fall through spring. During this period levels in Systems 3 and 4 were predominantly between 5 and 10 mg/L. In winter, monthly average concentrations slightly above 15 mg/L occurred occasionally in the effluent of all systems except System 4. This may have been due to shortened retention times resulting from ice blockages. Elevated concentrations of suspended solids occurred in one or more years in all systems in summer or late spring due to phytoplankton blooms. This was compounded by the concentration of suspended material resulting from evapotranspiration.

Although suspended solids levels were, on average, five times as high in the aeration cell effluent than in the lagoon effluent, in most seasons comparable concentrations of suspended solids were measured from Channel C to the ends of Systems 3 and 4 (Figure A.3). During the summer of 1983, characterized by high evaporation rates and stagnation, suspended solids increased as the wastewater circulated through the channels, particularly in System 4. In the summer of 1984, suspended solids were lower in all channels, although influent concentrations were considerably higher.

The four year average effluent concentrations were lowest in System 4 (8.0 mg/L), followed by Systems 1, 2, 3 and 5 at 8.6, 9.0, 9.2 and 11.6 mg/L, respectively (Table 5.4). Although System 3 achieved lower average effluent concentrations then System 1 and 2 (in 3 out of 4 years), high algal concentrations in the summer of 1983 increased the 4 year average. Annual removal efficiency ranges were 34-70%, 28-71% and 40-75% in Systems 1, 2 and 3, respectively (Table 5.5). Systems 4 and 5 showed efficiencies of 90-94% and 77-92%, respectively. Although the channelized systems performed best, the differences in treatment between systems receiving the same type of influent were not as great as with other parameters. This was a result of the importance of physical processes (e.g., sedimentation, filtration) in suspended solids removal as opposed to processes dependent on biological transformations that are more sensitive to the conditions created by alterations in retention time and flow distribution.

5.2.2 Biological Oxygen Demand

BOD_5 levels are a measure of the unstable organic matter (i.e., soluble and readily hydrolyzed forms) normally present in domestic sewage which, through aerobic decomposition, oxidizes (in 5 days) to stable inorganic forms. The BOD_5 provides an indication of the potential effect of these substances on the dissolved oxygen concentration in water.

BOD_5 levels in the aeration cell effluent were high (samples ranging from 10 to 168 mg/L) although some BOD reduction was achieved. BOD_5 in the lagoon effluent was considerably lower (samples ranging from 3 to 29 mg/L) due to settling and decomposition that occurred during treatment in the lagoon. Both types of influent fed to the marsh cells often showed winter and/or summer peaks in BOD_5 levels.

Table 5.4 Annual Average Suspended Solids Concentration (mg/L) in Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	22.8	13.2	14.6	13.0	119.2	9.6	20.2
1981/82	26.3	6.6	8.3	5.5	105.6	5.8	7.2
1982/83	23.5	8.4	6.5	13.3	110.3	10.4	10.9
1983/84	18.7	6.2	6.5	4.8	109.2	6.1	7.9
4 YR.AVG.	22.8	8.6	9.0	9.2	111.1	8.0	11.6

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5.

Table 5.5 Percent Removal of Suspended Solids Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	33.8	27.6	40.3	89.6	82.8
Fall	66.6	57.4	54.1	97.9	97.6
Winter	-7.1	8.0	15.4	87.0	82.0
Spring	48.8	43.8	59.7	82.1	74.0
Summer	35.5	3.2	16.7	90.7	77.2
1981-82	69.9	61.6	74.9	93.2	91.3
Fall	72.5	64.1	74.8	95.9	94.6
Winter	57.0	56.8	80.3	97.5	88.9
Spring	64.7	51.9	71.4	90.0	92.0
Summer	79.9	70.7	75.1	88.8	85.9
1982-83	63.3	70.5	53.3	93.4	90.4
Fall	-14.7	62.3	50.3	97.2	92.9
Winter	86.2	84.5	87.4	94.3	88.0
Spring	86.4	79.2	74.1	96.6	90.9
Summer	8.5	25.6	-67.0	84.3	88.6
1983-84	62.1	66.8	73.4	94.2	92.4
Fall	-5.3	39.3	48.1	96.1	95.7
Winter	64.4	22.5	57.9	92.1	67.4
Spring	74.4	81.9	84.5	97.1	97.0
Summer	75.6	88.4	81.1	91.6	98.3
4 Year Avg.	57.3	56.6	60.5	92.6	89.2

Monthly average BOD_5 concentrations and seasonal loadings are illustrated in Figures A.4 and A.5, respectively. Marsh effluent BOD_5 levels were generally less then 15 mg/L during the ice-free months, often between 5 and 10 mg/L. In the period of January through March, the monthly average BOD_5 rose in some years to maximum levels of 30 mg/L, 26 mg/L and 23 mg/L in Systems 1, 3 and 4, respectively, and to 50 mg/L in Systems 2 and 5. In winter, low temperatures result in reduced microbial activity and, often, to oxygen depletion under the ice-cover, both contributing to reduced rates of organic decomposition. However, winter BOD_5 levels in System 3 effluent above 15 mg/L occurred in March only and in only 2 of the 4 years.

Increases in BOD_5 levels occurred occasionally in one or more summer months, reaching maximum monthly concentrations of 24 mg/L, 32 mg/L, 34 mg/L and 35 mg/L in Systems 2, 3, 4 and 5, respectively. Monthly averages greater than 15 mg/L, when they occurred, were maintained for more than one summer month in Systems 4 and 5 only. The elevated summer BOD_5 levels were in most cases coincident with increases in suspended solids concentrations.

The pattern of changes in BOD_5 levels as the wastewater circulated through the channelized systems was similar to changes observed for suspended solids (Figure A.6) suggesting that settling of easily hydrolyzed organic particles represents at least part of the BOD_5 removal mechanism. Following the initial decline observed in Channel B there was a gradual reduction in BOD_5. Summer BOD_5 increases in the channels were probably a result of the concentrating effect of evapotranspiration and organic matter released from the lysis of dead algal cells.

Annual average BOD_5 levels were consistently lower in the channelized systems (3 and 4). Concentrations in the effluent of System 1 (which contained a channelized marsh plus a pond and open marsh) demonstrated annual BOD_5 concentrations only slightly higher than those in System 3 (Table 5.6). Annual treatment

efficiencies ranged between 50-66%, 20-56% and 56-63% in Systems 1, 2 and 3, respectively, and between 78-88% and 61-77% in Systems 4 and 5, respectively (Table 5.7). Seasonal removal efficiencies in System 3 were greater than 40% except in the first summer when the hydraulic loading was elevated and in the third summer when severe oxygen deficits were measured in the marshes. BOD_5 removal efficiencies remained high in winter (68-85%). Treatment efficiencies often declined in March when oxygen levels were low.

5.2.3 Dissolved oxygen and sediment redox potential

The maintenance of dissolved oxygen in water circulating through a marsh treatment system is advantageous for wastewater renovation because an oxygen supply 1) is required for the transformation and removal of toxic substances such as ammonia, 2) influences the rate of decomposition and removal of organic matter in the wastewater and 3) controls the release and immobilization of substances from the sediment.

The Listowel marsh systems demonstrated periods of high dissolved oxygen levels (monthly averages up to 13 mg/L) in spring and fall and varying degrees of oxygen depletion in the summer and winter (Figure A.7). These oxygen fluctuations are similar to those observed in undisturbed natural marshes. The high productivity of the marsh ecosystem, plus the organic matter loadings associated with the wastewater, results in high levels of bacterial decomposition and oxygen consumption, particularly at elevated summer temperatures. In winter, the ice cover acts as an impediment to diffusion of oxygen from the atmosphere and reserves of oxygen in the wastewater may be depleted. Low water volumes in marshes provide a limited reserve of dissolved oxygen and relatively slow diffusion through water limits the rate of oxygen replenishment.

Table 5.6 Annual Average BOD_5 Concentration (mg/L) in the Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	22.0	10.4	13.9	9.5	67.7	12.3	17.8
1981/82	22.8	9.5	15.5	8.4	53.6	8.8	16.7
1982/83	16.2	5.6	7.0	6.7	53.0	8.0	11.1
1983/84	17.2	6.6	8.9	5.9	50.7	9.2	12.9
4 YR. AVG.	19.6	8.0	14.8	7.6	56.3	9.6	14.6

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5

Table 5.7 Percent Removal of BOD_5 Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	50.00	25.3	57.2	77.5	70.5
Fall	62.8	72.3	74.8	96.0	95.6
Winter	33.0	7.4	56.7	77.2	72.3
Spring	62.8	36.4	61.8	67.6	54.4
Summer	44.8	16.5	34.6	77.3	65.6
1981-82	49.7	20.3	55.7	80.3	61.3
Fall	57.9	4.2	42.5	85.2	70.7
Winter	48.7	16.7	67.6	84.5	49.8
Spring	42.9	25.4	47.7	73.5	64.4
Summer	59.8	28.0	55.0	75.5	71.9
1982-83	65.6	56.0	62.9	87.6	76.7
Fall	55.3	53.0	69.1	92.2	83.0
Winter	83.4	64.7	84.7	91.5	73.4
Spring	83.5	72.4	81.0	94.6	84.9
Summer	19.3	17.2	-3.9	67.7	61.2
1983-84	55.2	46.6	61.9	80.8	73.2
Fall	-5.6	42.5	61.4	79.1	66.0
Winter	71.8	17.0	72.2	81.3	61.0
Spring	60.1	47.1	42.9	79.7	75.2
Summer	65.4	82.4	75.7	82.7	85.9
4 Year Avg.	55.1	37.1	59.4	81.6	70.4

Although oxygen is released by algae during daylight hours, this source was restricted in summer by shading from cattail stalks and duckweed growth on the water surface. Reports indicate that rooted emergent marsh plants are also a source of oxygen. Cattails transport oxygen downward to the roots where the excess, not used in root cell respiration, is released. However, the oxygen released from the plants may be readily utilized by microbes in the rhizosphere.

The lagoon effluent contained the highest monthly average oxygen levels in late fall (maximum 14 mg/L) with secondary peaks in the spring (maximum 11 mg/L). Sample levels ranged between 0 and 23 mg/L, the latter supersaturated condition resulting from algae growth in the lagoon. Monthly average concentrations in the lagoon effluent declined in winter (close to or at 0 mg/L) and in summer (generally greater than 2 mg/L). Oxygen levels in marsh systems 1-3 in most years showed similar seasonal fluctuations, although generally lower than influent levels in spring, fall and summer but higher in winter. During the first year oxygen levels in marsh Systems 1-3 exceeded influent levels in all seasons except summer. In the succeeding years influent levels were exceeded in winter only. In System 3, oxygen levels did not show the steep decline to zero in winter that often occurred in the influent but the data indicate gradual decline in oxygen from early winter into summer which was sometimes broken by elevated spring levels. Excluding the first year, monthly oxygen levels were highest in System 3 (maximum 8.5 mg/L), followed by System 1 (maximum 7.5 mg/L), with considerably lower levels in System 2 (maximum 4.0 mg/L). During the 2nd and 3rd year, the monthly average dissolved oxygen concentration in System 2 effluent did not rise above 2.0 mg/L.

Oxygen levels in the aeration cell effluent were variable with monthly oxygen concentrations reaching a maximum of 6.0 mg/L. Although the levels were generally lower than in the lagoon effluent, they were not as depressed as the latter during the period of ice cover. The dissolved oxygen concentration in the aeration cell tended to be lowest in summer. This may be related to the

reduced solubility (40% between 0^{o} and $25^{o}C$) of oxygen and increased biological activity at high temperatures. Dissolved oxygen levels in the aeration cell ranged between 0. and 8.0 mg/L.

Oxygen concentrations in Systems 4 and 5 were elevated in fall and/or spring but, in most years, were lower than in systems of the same design receiving lagoon effluent (Systems 3 and 2). During the first year the oxygen concentration in the effluent of Systems 4 and 5 greatly exceeded influent levels in spring and fall and were comparable with levels in the other marsh systems. In the following 3 years the oxygen concentrations in Systems 4 and 5 (monthly maximum 5.5 mg/L) were lower than in System 1 and 3. Dissolved oxygen concentrations were considerably lower in the effluent of System 5 than in System 4.

During most months the channelized systems (3 and 4) demonstrated higher oxygen concentration than the other systems receiving the same type of influent. The placement of air blowers in the culverts between the channels to prevent ice blockage in both systems in winter, beginning in the second year, may have contributed to higher oxygen levels in winter. System 3 maintained the highest levels during the 4 years as a result of both higher influent oxygen levels and lower BOD when compared with System 4.

The nutrient status of overlying waters is strongly influenced by the presence of oxygen in the water and the upper layer of the sediment (Howeler, 1972). When the oxygen supply is limited, a proportion of soil microoganisms make use of electron acceptors other than oxygen for the respiratory oxidation of organic matter. This results in the conversion of numerous compounds to a state of chemical reduction and is reflected in the lowering of the oxidation-reduction (redox) potential. The redox potential of a system (Eh) is a measure of the tendency to receive or supply electrons and is governed by the nature and proportions of oxidizing and reducing substances it contains (Yoshida, 1975).

Redox potentials are used to characterize negative soil aeration and potentials have been defined at which a number of important chemical changes occur (Reddy, 1985). The most efficient carbon removal from wastewater occurs during aerobic respiration at redox potentials >+300 mV. The redox potential + 300 mV is considered by Reddy (1985) to represent the boundary between oxidized and reduced conditions. Facultative anaerobic respiration predominates between - 100 and +300 mV. Under these conditions characterized by the absence of oxygen, bacteria utilize nitrate, oxidized manganese compounds and ferric iron compound as election acceptors (in that order). As a consequence, nitrates are reduced to nitrous oxide and nitrogen gas and the metals are converted to their reduced (and more soluble) forms. In the absence of the above electron acceptors, true anaerobes utilize sulfate and carbon dioxide at redox potentials between -300 and -100 mV, forming sulfides and methane.

The presence of the oxidized sediment surface acts as a barrier to the products of anaerobic respiration formed in the underlying sediments. However, during periods of high oxygen consumption or inadequate supply, the oxidized sediment surface thins and may disappear, thereby permitting the movement of reduced forms of nitrogen, iron, manganese, carbon and sulfur and adsorbed ions such as phosphate and ammonium, into the overlying water (Mortimer, 1941).

The pattern of changes in sediment redox potential in Systems 2, 4 and 5 over a 2 year period are illustrated in Figure 5.13. A decline in redox potential around or below - 100 mV was observed in the summer of both years whereas the decrease in redox potential in winter was not as severe.

Redox potentials were measured at 1.5 cm depth in the sediment of Channels A (stns. 1, 2 + 3), B (stns. 4, 5 + 6) and C (stns. 7, 8 + 9) in Systems 3 + 4 during the ice-free period (Mar.-Nov.) in 1983 and 1984. The range of sediment redox measurements and mean in each season is listed in Table 5.8. Figure A.8 illustrates the monthly average sediment redox potential at consecutive stations in the first, third and fifth (last) channel in Systems 3 and 4.

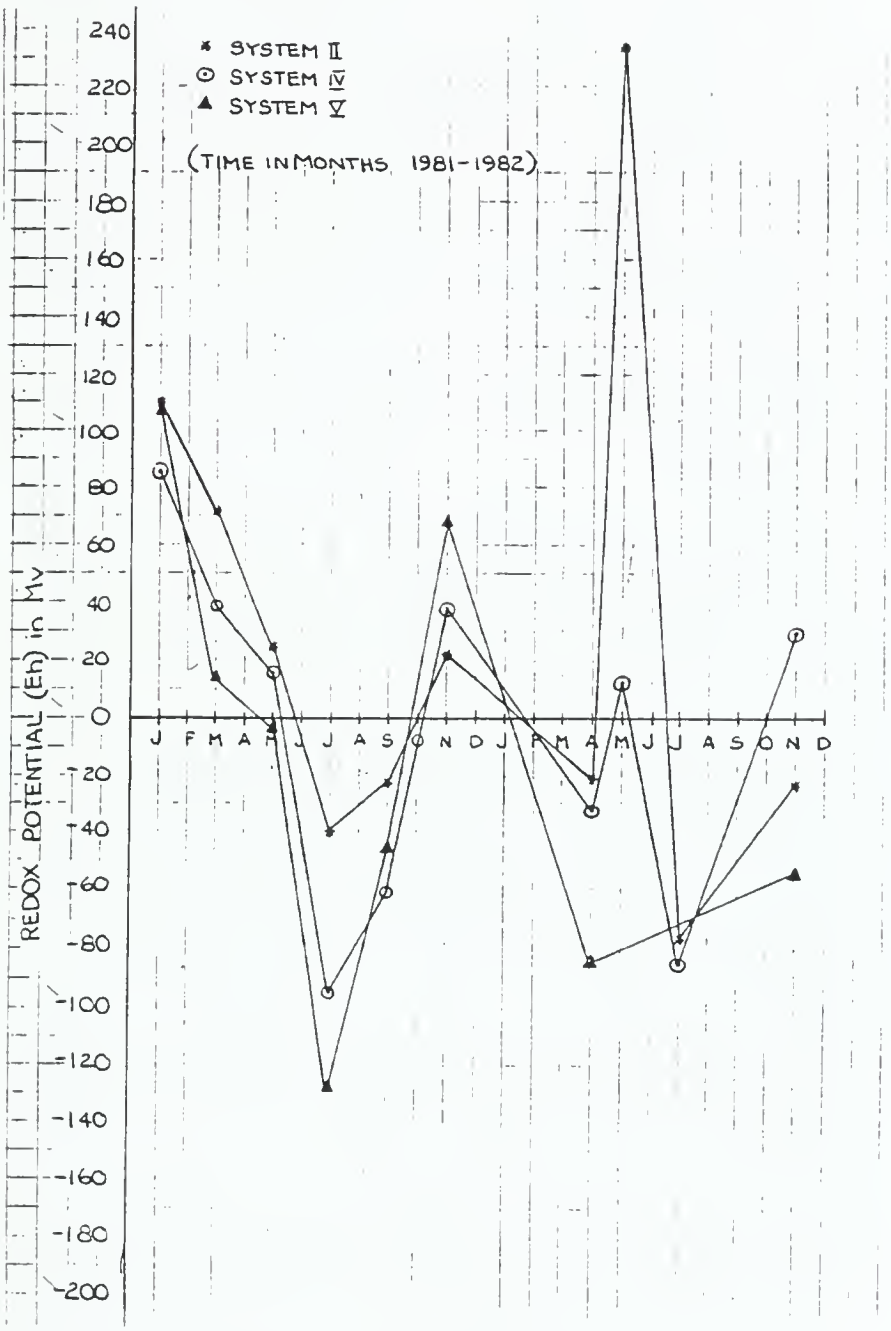


Figure 5.13 Average redox potential in sediment measured at the sediment bacteria sampling stations at influent and effluent ends of Systems 2, 4 and 5 during two year period (1981 and 1982).

Redox potentials were slightly lower in System 4 then in System 3 in all ice-free seasons except the summer of 1983. Greater differences were apparent between years. Although the seasonal mean potential (Table 5.8) in both years were similar in spring (ranging between -74 and -49 mV), redox potentials differed between years (1983 and 1984) in summer (-160 mV vs. +2mV in System 3 and -143 mV vs. -28 mV in System 4) and the differences followed over into the fall (-85 mV vs. -4 mV in System 3 and -97 mV vs. -11 mV in System 4). Minimum redox potentials measured in Systems 3 and 4 were -319 mV and -284 mV, respectively, in the summer of 1983. The difference between the latter two measurements may not he significant as the accuracy in similar measurements done elsewhere (Jone, 1979) was estimated to be within 50 mV.

No sediment redox potentials greater than + 300 mV were measured at the depth measurements were taken (1.5 cm). This does not preclude the presence of an oxidized zone at the sediment surface during periods when dissolved oxygen was present in the overlying water.

In most months the redox potential increased (to varying extents) between Stations 1 and 9, particularly noticeable in the spring months. In 1984, each summer month had progressively lower redox potentials whereas in 1983 it was lowest in July and showed a rebound in August (close to the June measurements in System 3).

An examination of mass balances in System 3 indicated atypically low removal efficiencies for several parameters in the summer of 1983 as compared with the summer in other years, i.e. soluble phosphorus: -232% vs. 34 to 84%, total phosphorus: -32% vs. 57 to 81%, SS: -67% vs. 17 to 81%, BOD_5: -4% vs. 35 to 76%, TKN: 36% vs. 46 to 78%, NH3: 46% vs. 50 to 86%. Redox potentials in the marshes were lowest (around - 300 mV) during this period. The mass balances preclude the concentrating effect of high evapotranspiration rates. The reduced phosphorus removal efficiencies may be related to the

Table 5.8 Redox Potential in Channels of Systems 3 and 4*

	System 3				System 4			
	1983		1984		1983		1984	
Season	Mean (mV)	Range (mV)	Mean (mV)	Range (mV)	Mean (mV)	Range (mV)	Mean (mV)	Range (mV)
Spring	-49	(-194 to + 85)	-52	(-164 to + 166)	-61	(-137 to + 83)	-74	(-131 to + 74)
Summer	-160	(-319 to -36)	+2	(-64 to + 110)	-143	(-284 to - 4)	-28	(-77 to + 48)
Fall	-85	(-152 to -7)	-4	(-58 to + 31)	-97	(-138 to -24)	-11**	(-60 to + 42)**

* Measurements taken at 2 week intervals at 9 stations in the sediment of Systems 3 & 4 during the ice-free period.

** Sept. and Oct. data only.

desorption of phosphates from the surface of hydrated oxides of iron which has been observed by researchers at low redox potentials (Snow and DiGiano, 1976). The lower rate and efficiency of microbial decomposition of organic substances under anaerobic conditions may be reflected in the decline in BOD_5, TP and TKN removal efficiencies.

5.2.4 Soluble reactive phosphorus

Soluble reactive phosphorus consists primarily of orthophosphates, although some readily hydrolyzed dissolved forms of phosphate may also be included in the measurement of this parameter (MOE, 1979). Orthophosphate is a form of phosphorus which is directly available to aquatic plants and, because it is often a limiting factor, controls the growth of algae in surface waters.

Monthly average soluble phosphorus concentrations are illustrated in Figure A.9. Soluble phosphorus levels in the aeration cell effluent and lagoon effluent were similar in the winter and spring, often showing small peaks in winter but remaining <0.4 mg/L. Concentrations in both types of influent rose above this level in the summer and early fall in all years. During the first summer and early fall when alum treatment was suspended, monthly average influent levels rose to the highest levels reached in the four years, 2.5 mg/L and 1.5 mg/L in aeration cell effluent and lagoon effluent, respectively. Soluble phosphorus concentrations were also elevated in the aeration cell effluent when alum was withheld during the second summer.

Marsh effluent soluble phosphorus levels were generally <0.4 mg/L (often <0.2 mg/L in spring and fall) with occasional small increases in winter or early spring, coincident with reduced dissolved oxygen levels. Effluent concentrations in all systems rose markedly in summer, and in some instances, through early fall.

Excluding the first two summers, monthly summer effluent levels increased to 1.6, 1.2 and 1.2 mg/L in Systems 1, 2 and 3, respectively. Soluble phosphorus summer peak concentrations in the effluent of systems receiving aeration cell effluent (Systems 4 and 5) were generally higher than maximum levels reached in Systems 1-3. Maximum monthly levels in Systems 4 and 5 during the last two years were 2.2 and 2.5 mg/L, respectively, the latter occurring in September. In the first summer (1981) when there were no alum additions, System 5 effluent levels remained high even when influent levels in September through November dropped to <0.1 mg/L. The highest summer soluble phosphorus levels occurred in the non-channelized systems. The four year average soluble phosphorus concentration was lowest in the effluent of System 3 (.26 mg/L), followed by System 4 (.34 mg/L) (Table 5.9).

Seasonal and annual soluble phosphorus loadings and removal efficiencies are illustrated in Figure A.10. Annual removal efficiencies ranged between -87 and 50%, - 98 and 34%, and -12 and 60% in Systems 1, 2 and 3, respectively (Table 5.10). Systems 4 and 5 demonstrated annual removal efficiencies of -138 to 59% and -245 to 40%, respectively. The data indicate a decline in annual treament efficiency in all systems. The decline was particularly steep in Systems 4 and 5. Whereas annual removal efficiency in System 3 stabilized around -10% in the last 2 years, the removal efficiency in System 4 fell to -57% and -138%, respectively, during the same period. Removal efficiencies were often low in winter and early spring, the lowest monthly efficiency in System 3 occurring in March in 3 out of 4 years. Although summer removal in System 3 was high in some years, as a result of assimilation by bacteria, algae and marsh plants, the removal efficiency was depressed (to -232%) in the summer of 1983 when anoxic conditions were prevalent in the marshes.

Table 5.9 Annual Average Soluble Phosphorus Concentration (mg/L) in Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	.54	.23	.33	.13	.64	.19	.36
1981/82	.48	.58	.75	.42	.64	.41	.84
1982/83	.23	.47	.46	.26	.18	.37	.24
1983/84	.22	.26	.25	.21	.15	.37	.52
4 YR. AVG.	.37	.39	.45	.26	.40	.34	.49

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5.

Table 5.10 Percent Removal of Soluble Reactive Phosphorus Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	49.6	33.6	59.2	58.7	40.1
Fall	85.9	92.5	96.0	95.5	88.5
Winter	-3.8	5.6	9.3	-12.6	19.2
Spring	53.8	36.4	16.2	66.8	56.1
Summer	49.3	18.7	79.3	65.7	35.1
1981-82	-32.2	-69.6	0.5	37.0	-82.1
Fall	-51.6	-147.3	2.6	-380.8	-1961.6
Winter	-102.0	-76.7	-58.6	-135.6	-212.0
Spring	-44.2	-51.4	-61.6	45.1	-14.7
Summer	17.8	14.4	33.6	73.5	75.3
1982-83	-86.5	-98.1	-12.3	-57.2	-43.9
Fall	35.6	-12.1	41.3	20.0	-65.5
Winter	-71.6	-136.3	-21.6	-57.7	-193.0
Spring	-81.0	-15.7	26.1	-17.6	-45.6
Summer	-577.4	-449.8	-231.5	-81.8	-10.0
1983-84	-24.5	-17.8	-10.4	-137.6	-245.3
Fall	-0.5	-28.6	-24.2	-181.8	-518.6
Winter	-74.0	-167.7	-115.4	-326.6	-503.6
Spring	-79.1	-76.5	-130.7	-108.5	-187.2
Summer	-1.8	74.8	84.1	-98.6	-105.3
4 Year Avg.	-23.4	-38.0	9.3	-24.8	-82.8

Soluble phosphorus in marshes is immobilized by adsorption and precipitation reactions with aluminum, iron, calcium and clay minerals (Nichols, 1983). Chemical adsorption was shown to dominate in soils in contact with solution concentrations of phosphorus up to 1 mg/L, above which physical adsorption predominated. Similar adsorption and precipitation processes may be involved in phosphorus retention in organic soils as studies suggest that phosphorus fixation in organic soils is related to the soils' ash, iron, aluminum and calcium content (Nichols, 1983). Immobilization of phosphorus by bacteria associated with decomposing litter in summer has been documented in this report. Although bacterial activity and the decomposition and release of organically bound phosphorus is high in warm weather, phosphorus uptake by bacteria and plants may explain the relatively high treatment efficiencies observed in summer in most years. Even though the major source of plant nutrients for emergent vegetation (such as cattail plants) is the marsh sediment rather than the overlying water (Sculthorpe, 1967), phosphorus is assimilated by algae and duckweed growing in the marsh, as well as by cattail roots exposed to the water. Chlorophyll a in System 3 effluent ranged between 14 and 80 ug/L in the summer months.

Adsorbed phosphorus is released from the sediment if phosphorus levels in the water are lowered, thereby changing the phosphorus equilibrium between the water and the sediment (Ellis, 1973). In addition, increases in phosphorus release from sediment have been observed with declining redox potential in the range -200 mV to 200 mV (Wetzel, 1975; Patrick and Khalid, 1974). This suggests an association with the reduction (and increased solubility) of ferric compounds which occurs in the same redox range. In controlled studies, the occurrence of increasing phosphorus solubility with decreasing pH was demonstrated in reduced sediments (De Laune et al, 1981).

The decline in soluble phosphorus removal efficiency over the four years suggests a reduction in the phosphorus adsorption capacity of the sediment. The decline in annual soluble phosphorus removal in Systems 3 and 4 with increases in the cumulative soluble phosphorus loadings is illustrated in Figure 5.14. Significant reductions in soluble phosphorus removal began as early as the second year and though initially steeper in System 3, the removal rate became severely depressed in System 4 by the fourth year. The high phosphorus loadings to all systems in the first year must have contributed to these early indications of a reduction in phosphorus retention. The apparent generation of large amounts of soluble phosphorus in System 4 was probably a result of the high total phosphorus loadings from the aeration cell effluent, as the soluble phosphorus loadings were similar in Systems 3 and 4 (Table 5.11). The sludge mats in Systems 4 and 5 contained large amounts of organically bound phosphorus which upon decomposition could have been a source of phosphates.

An investigation of phosphorus removal efficiencies in different types of wetlands (e.g. log, fen, swamp) tested as wastewater filtration systems indicated phosphorus saturation in a few years, followed by phosphorus export (Richardson, 1985). The phosphorus adsorption capacity was found to be related to cumulative phosphorus loadings and could be predicted solely from the extractable aluminum content of the soil.

Studies indicate that phosphorus adsorption is dependent on the chemical and physical properties of the sediment. Although the mineral and clay content of the sediment in the Listowel artificial marsh systems was relatively stable due to the absence of stormwater drainage in the systems, the redox potential of the sediment varied considerably during the year. The extended periods in summer and winter with low dissolved oxygen levels probably contributed to low phosphorus retention as adsorption onto oxyhydroxides of iron and aluminum is believed to be one of the ways in which phosphorus is retained in sediments (Hook et al, 1973).

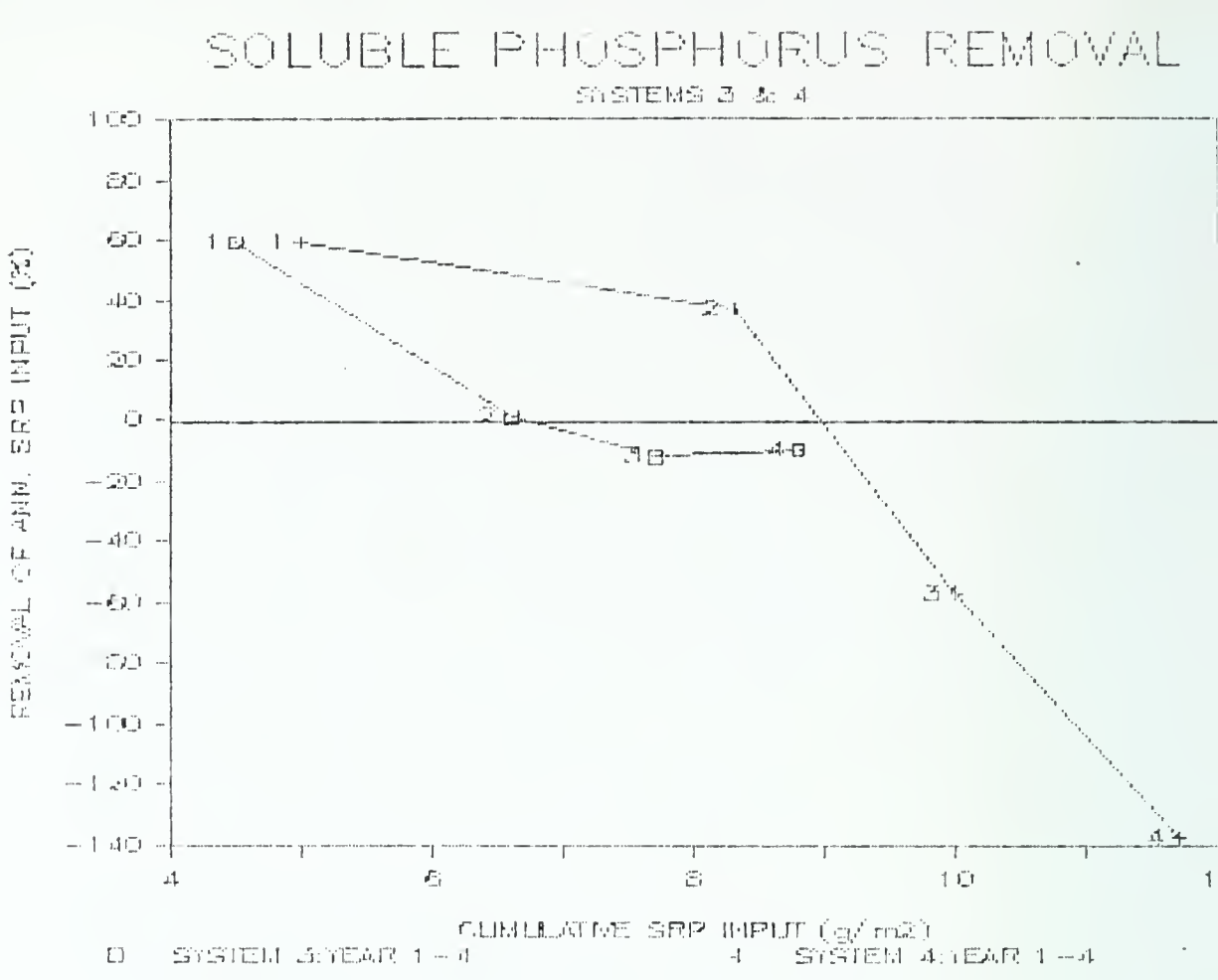


Figure 5.14 Cumulative soluble phosphorus loadings and removal rates in Systems 3 and 4 (4 years).

Table 5.11 Annual Phosphorus Loadings

Soluble Reactive Phosphorus (g/m^2)

YEAR	SYSTEM				
	1	2	3	4	5
1	5.5	6.3	4.5	5.0	6.4
2	3.5	3.2	2.1	3.3	2.8
3	1.7	1.7	1.1	1.1	1.4
4	1.7	1.7	1.1	1.2	1.6

Total Phosphorus (g/m^2)

YEAR	SYSTEM				
	1	2	3	4	5
1	15.5	20.0	14.4	33.2	46.3
2	9.8	9.0	6.1	15.4	15.4
3	6.2	5.9	3.9	16.2	21.0
4	6.1	6.0	4.0	20.6	27.1

5.2.5 Total phosphorus

Weekly total phosphorus concentrations ranged between 0.8 and 9.1 mg/L in aeration cell effluent and between 0.3 and 2.4 mg/L in the lagoon effluent. The fluctuations in total phosphorus levels in the aeration cell effluent were very similar to those of the suspended solids concentrations indicating that most of the total phosphorus was associated with the suspended particles. On average only 13% of the total phosphorus in aeration cell effluent was soluble due to alum pre-treatment, considerably lower than in the lagoon effluent (34%). Total phosphorus in the lagoon effluent was mainly below 1 mg/L. Exceptions occurred primarily in summer and early fall when total phosphorus rose to a maximum slightly above 2 mg/L. Levels were highest in both types of influent from May to October, 1981 due to the suspension of alum treatment during that period.

Monthly average total phosphorus concentrations and seasonal loadings are illustrated in Figures A.11 and A.12, respectively. Marsh effluent levels were generally below 1 mg/L and often below 0.5 mg/L in the period of fall through spring. Small peaks in the monthly averages at the end of winter (usually March) slightly exceeded 1 mg/L in Systems 3 and 4 in the second year whereas these were more pronounced in Systems 1 and 2 and extended over several months in System 5, reaching 1.5 mg/L. Elevated total phosphorus levels occurred in summer (usually July) in one or more years in all systems.

However, in systems receiving lagoon effluent (Systems 1-3), monthly average values exceeded 1 mg/L in 2 out of 4 years, during the summer of 1981 when alum treatment was suspended and in summer 1983 when weather conditions contributed to severe oxygen depletion. Maximum summer effluent levels in System 3 were 1.2 mg/L in July, 1982 and 1.8 mg/L in July, 1983. In contrast, systems receiving aeration cell effluent (Systems 4 and 5) exceeded 1 mg/L, often for 2 summer months (July and August), in all years. The maximum monthly level in System 4 was 2.9 mg/L.

The three major forms of phosphorus in raw domestic sewage are soluble orthophophate, soluble polyphosphate and suspended particulate phosphorus. Alum (aluminum sulfate) converts soluble orthophosphate to insoluble forms through reactions that include the formation of aluminum phosphate precipitates. The liquids-solids separation, the second step in the process of chemical phosphorus removal, occurs during sedimentation. Seasonal concentrations in the channels of Systems 3 and 4 demonstrate the difference in total phosphorus concentrations in the two systems and the pattern of change in concentration as the wastewater circulates through the channels (Figure A.13).

Although, on average, 50% of the phosphorus in the marsh system effluents was soluble phosphorus, a major part of the total phosphorus concentration in summer was usually in the soluble fraction. Several factors contributed to the elevation in total phosphorus concentration in summer. Greater bacterial activity at high temperatures increases the mineralization of phosphorus from organic matter and oxygen depletion results in reduced sediment redox potentials which have been associated with phosphorus release from the sediment. The higher total phosphorus concentrations in Systems 4 and 5 may reflect the "pool" of insolubles phosphates which sedimented out following release from the mixed aeration cell.

System 3 demonstrated the lowest total phosphorus effluent over the four year period (0.5 mg/L), followed by System 4 (0.6 mg/L), although average influent levels in System 4 were more than three times the levels in System 3 influent (Table 5.12). Annual treatment efficiencies ranged between -11 and 49%, -1 and 40% and 39 and 53% in Systems 1, 2 and 3, respectively (Table 5.13). Systems 4 and 5 demonstrated annual efficiencies of 76-84% and 43-79%, respectively. The highest treatment efficiencies often occurred in summer or spring probably indicating the influence of bacterial and plant uptake at this time of year. The decline in the soluble phosphorus annual removal efficiency discussed previously was not apparent in the case of total phosphorus in any of the marsh systems.

Table 5.12 Annual Average Total Phosphorus Concentration (mg/L) in Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	1.3	.60	.74	.48	3.8	.62	.98
1981/82	1.3	.94	1.2	.71	3.2	.73	1.6
1982/83	.82	.68	.66	.44	2.9	.55	.56
1983/84	.76	.47	.43	.38	2.8	.58	.82
4 YR. AVG.	1.1	.67	.76	.50	3.2	.62	.99

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh system 1-5.

Table 5.13 Percent Removal of Total Phosphorus Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	49.0	33.5	52.5	78.6	70.5
Fall	80.3	84.6	87.3	97.6	96.5
Winter	22.9	13.2	25.9	71.7	64.1
Spring	50.6	37.5	54.4	73.4	67.8
Summer	53.1	29.1	66.5	80.6	65.4
1981-82	20.1	-0.7	39.2	75.5	43.1
Fall	-10.5	-71.9	23.7	69.7	3.5
Winter	6.8	-3.9	31.4	71.0	50.1
Spring	30.3	16.2	37.3	75.4	37.7
Summer	44.9	45.0	57.1	81.4	80.3
1982-83	21.8	17.7	47.2	83.8	79.3
Fall	43.9	22.1	53.8	91.9	83.3
Winter	49.0	28.8	61.2	85.7	66.2
Spring	69.2	66.0	81.9	94.9	87.4
Summer	-126.3	-76.3	-32.3	64.1	75.6
1983-84	35.8	40.4	45.2	78.4	69.0
Fall	26.8	23.0	28.2	76.7	54.4
Winter	23.3	-6.7	11.9	76.4	60.0
Spring	42.4	43.0	41.8	82.6	77.0
Summer	45.8	79.7	81.2	77.8	78.0
4 Year Avg.	31.7	22.7	46.0	79.1	65.5

5.2.6 Ammonia

Ammonia levels in the aeration cell effluent (range 1.2-19.7 mg/L) were slightly higher than in the lagoon effluent (range 0.1-16.5 mg/L). Four year mean values were 8.6 and 7.2 mg/L, respectively. Peak concentrations occurred in both types of influent during summer and winter; minimum levels were observed in the spring and/or fall. Monthly average ammonia concentrations and seasonal and annual loadings are illustrated in Figures A.14 and A.15, respectively.

The pattern of ammonia levels in the marsh effluents was similar to that in the influents, although marsh effluent concentrations were lower in most months. Monthly effluent levels declined to $\leq$1.0 mg/L for varying lengths of time in the spring and early summer in all systems receiving lagoon effluent (Systems 1-3).

Ammonia levels in the effluent of Systems 4 and 5 decreased to $\leq$3.0 mg/L and $\leq$7.0 mg/L, respectively, during this period. Minimum fall levels were generally not as low as levels measured in spring. Minimum monthly levels in the fall ranged from $\leq$1 mg/L to 5 mg/L and 10 mg/L, respectively, in systems receiving lagoon effluent and aeration cell effluent.

Elevated ammonia levels were observed in the summer and winter in all marsh systems. The effluent of Systems 2 and 3 reached monthly average concentrations up to 13 mg/L whereas in Systems 4 and 5 levels rose to 16 mg/L and 18 mg/L, respectively. Systems 1 and 3 demonstrated considerable ammonia removal in summer and winter, less so in System 2. The systems receiving aeration cell effluent showed little ammonia removal during influent peaks in summer and winter. System 4 and 5 effluent levels often exceeded influent levels at these times. System 5 effluent exceeded monthly influent levels for several consecutive months in the fall, winter and summer.

Nitrification is a major mechanism of ammonia removal in marshes. It is carried out by aerobic autotrophic bacterial populations that sequentially oxidize ammonia to nitrate with the intermediate formation of nitrite. These organisms, the genera Nitrosomonas and Nitrobacter, derive energy for growth from the oxidation of inorganic nitrogen compounds rather than from organic compounds.

Ammonia and substances yielding ammonia (i.e. organic nitrogen compounds) exert an oxygen demand in addition to that exerted by organic (carbon) compounds as measured in the BOD_5. The oxygen consumption reactions associated with ammonia are described in the following equations:

1. $NH_4^+ + 1.5\ O_2 \xrightarrow{\text{Nitrosomonas}} 2H^+ + HO_2 + NO_2^-$

2. $NO_2^- + 0.5\ O_2 \xrightarrow{\text{Nitrobacter}} NO_3^-$

The oxygen consumption of the conversion of 1 gm ammonia nitrogen to nitrite is 3.43 gm O_2 whereas the conversion of 1 gm nitrite-nitrogen to nitrate is 1.14 gm O_2 (Semkin et al, 1976). The total theoretical oxygen utilization in the nitrification process is 4.57 gm of oxygen per gram of ammonia transformed to nitrate. Therefore, about 46 mg/L of oxygen is required to completely oxidize 10 mg/L ammonia. The oxygen demand associated with ammonia is considerably greater than the oxygen demand necessary for the stabilization of organic substances (0.5 - 1.5 gm O_2/gm BOD_5).

The examination of ammonia and dissolved oxygen concentrations over the four year period suggests that elevated ammonia levels in summer and winter may have resulted from different limiting factors. Dissolved oxygen levels in the marsh systems were considerably lower in summer than in the winter months. For example, minimum monthly average oxygen concentration in System 3 effluent over the study period ranged from 1.5 to 5.5 mg/L in winter whereas levels in

summer ranged between 0 and 1.5 mg/L (Figure A.7). March levels ranged between 0 and 4 mg/L. Oxygen levels in summer were low enough to inhibit nitrification but the oxygen supply in winter, although declining, was in some years sufficiently high to support nitrification. Summer oxygen deficits often extended into September. Because the process of nitrification is dependent on the activity of the bacterial nitrifiers, the limiting factor in winter, in some years, may have been the low winter temperatures. Nitrification has been reported to occur between 4^{o} and $45^{o}C$ (Sutton and Jank, 1976). Average monthly temperatures in the marsh systems were generally below $4^{o}C$ from December through March.

In the spring and fall the lagoon effluent contained elevated oxygen levels (often supersaturated) generated by algae growing in the lagoon. Summer oxygen levels rarely fell below 1 mg/L in the lagoon effluent but often fell to zero in the winter. Dissolved oxygen concentrations in the aeration cell effluent were generally lower, monthly average values were <1 mg/L in 2-3 months in and around the summer period, but often moderate oxygen supplies were maintained in winter.

The minimum dissolved oxygen required for nitrification has been controversial. Complete nitrification has been observed in treatment plants at dissolved oxygen levels as low as 0.5 mg/L. However, studies indicate a reduction in the rate of nitrification at low oxygen levels (EPA, 1975).

Another mechanism of ammonia removal is the adsorption of ammonium ions onto sediment particles in contact with wastewater circulating through the marsh. The cation exchange capacity (a measure of the negative charge) of organic and inorganic colloids provides sites for the adsorption of ammonium ions (Brady, 1974). Some of the charges on clays and all of the charges on organic colloids are considered pH-dependent. However, anaerobic conditions at the sediment-water interface have been reported to trigger the release of ammonia into the overlying waters (Mortimer, 1941).

Other ammonia removal mechanisms include assimilation by bacteria and marsh plants and volatilization of ammonia. Ammonia uptake by plants involves the conversion to organic forms of nitrogen and is temporary unless the vegetation is harvested. However, at least part of the decomposition of organic nitrogen may take place in a less sensitive season for receiving waters, or when other removal mechanisms are operating.

Volatilization of ammonia, which occurs at pH values greater than 7.5, may have provided some ammonia removal in the marsh systems. The monitoring data indicated pH effluent values rose above 7.5 for only a short time at the start of marsh operation. However, samples were taken early in the morning and does not preclude higher pH after midday due to algal photosynthesis.

Channel sampling in Systems 3 and 4 indicated gradual decreases in ammonia concentration (Figure A.16). The increase between the influent and Channel B in System 3 in the summer may have resulted from the concentrating effect of evapotranspiration. The decomposition of organic nitrogen compounds in the accumulated sludge may have contributed to the increase in Channel B of System 4 in all seasons.

Average ammonia concentrations over the four year period were lowest in System 3 (3.8 mg/L), followed by Systems 1, 2 and 4 with 4.9, 5.1 and 6.1 mg/L, respectively (Table 5.14). Average annual removal efficiencies ranged between 32 and 51% in System 3 and between 5 and 35% in System 4 (Table 5.15). In both systems removal efficiencies were lowest in winter; in System 4, all winter removal efficiencies were negative as compared to the range of 11 to 36% in System 3 (Figure A.15).

Table 5.14 Annual Average Ammonia and Un-ionized Ammonia Concentration in Marsh Systems

	AMMONIA CONCENTRATION (mg/L)						
YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	6.8	3.5	4.5	3.6	8.3	4.5	5.9
1981/82	7.7	6.7	7.3	4.6	8.5	6.0	10.3
1982/83	6.5	4.0	4.7	2.9	8.9	5.9	6.3
1983/84	7.6	5.2	3.9	4.0	8.6	8.1	9.2
4 YR. AVG.	7.2	4.9	5.1	3.8	8.6	6.1	7.9

	UN-IONIZED AMMONIA CONCENTRATION (mg/L)						
YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	.09	.02	.02	.02	.05	.01	.03
1981/82	.23	.03	.02	.01	.11	.01	.03
1982/83	.13	.01	.01	.01	.06	.04	.01
1983/84	.05	.01	.003	.003	.04	.01	.01
4 YR. AVG.	.13	.02	.01	.01	.07	.02	.02

SLBE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5.

Table 5.15 Percent Removal of Ammonia Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	40.3	20.1	30.7	27.6	21.1
Fall	98.3	97.7	96.5	96.0	93.6
Winter	14.0	11.1	11.2	-15.7	-7.9
Spring	49.6	11.6	26.9	43.7	42.0
Summer	48.2	26.9	54.7	38.9	7.0
1981-82	3.5	2.0	31.7	22.4	-47.8
Fall	-28.2	-64.5	40.3	2.4	-180.0
Winter	4.5	20.2	19.2	-16.4	-40.9
Spring	7.7	0.7	27.4	33.1	-67.1
Summer	22.2	28.7	50.2	55.9	58.5
1982-83	38.3	25.3	50.5	35.1	22.7
Fall	37.3	38.6	59.6	44.5	36.4
Winter	37.3	4.9	36.0	-13.6	-10.1
Spring	79.1	31.0	62.9	59.8	34.2
Summer	12.5	25.5	45.6	40.0	24.3
1983-84	25.9	45.3	42.2	4.9	-8.6
Fall	18.5	56.4	27.6	-9.8	-46.3
Winter	10.3	4.5	19.8	-20.7	-22.3
Spring	35.9	29.0	33.9	31.6	-10.3
Summer	41.4	89.2	86.0	13.1	24.2
4 Year Avg.	27.0	23.2	38.8	22.5	-3.2

The toxicity of ammonia to aquatic organisms is attributed to the un-ionized ammonia species (NH_3) present in aqueous solutions. The ammonium ion (NH_4^+) is considered less toxic, although objectionable in surface waters, due to its high oxygen demand (upon conversion to nitrates) and because of its nutritive properties which may promote excessive growth of aquatic plants. The proportion of un-ionized ammonia in solution rises with increases in pH and, to a lesser extent, with increases in water temperature.

Monthly average un-ionized ammonia levels in all marsh system effluents were maintained below .02 mg/L in most months during the four year period (Figure A.17). Slight increases occurred in the spring or summer reaching maximum monthly average concentrations of .07, .05 and .04 mg/L in Systems 1, 2 and 3, respectively, a substantial reduction from the high levels at this time in the lagoon effluent. Algae production in the lagoon causes increases in pH in spring and summer. These conditions, in combination with elevated temperatures, results in an increase in the proportion of un-ionized ammonia. In contrast, low temperatures and pH values under the ice cover in lagoon in winter, results in a decline in this parameter.

Although un-ionized ammonia levels were lower in the aeration cell effluent than in the lagoon effluent, effluents of systems receiving wastewater from the aeration cell (Systems 4 and 5) reached peak monthly un-ionized ammonia levels of .3 and .1 mg/L, respectively, in summer. These increases were influenced by the high accumulations of total ammonia under anaerobic conditions in summer.

5.2.7 Total Kjeldahl Nitrogen

Total Kjeldahl nitrogen concentrations were, on average, 1.6 times higher in the aeration cell effluent (7.0-37.0 mg/L) than in the lagoon effluent (5.0-26.5 mg/L). Nitrogen in raw domestic sewage is principally in the form of organic nitrogen (both soluble and particulate) and ammonia. Ammonia represented 46% of the annual average total Kjeldahl nitrogen concentration in the aeration cell effluent and 60% in the lagoon effluent. Seasonal trends were apparent in the elevated total nitrogen levels in both types of influent in the summer and winter.

Monthly TKN concentrations and seasonal loadings are illustrated in Figures A.18 and A.19, respectively. Total nitrogen concentrations in the marsh system effluents were in all months (except one) lower than influent levels but generally followed the influent pattern. During July, 1983, when high evapotranspiration rates caused severe stagnation, total Kjeldahl nitrogen levels in Systems 1, 2 and 4 exceeded influent levels. Effluent concentrations were lowest in the spring and fall ($\leq$10 mg/L, often $\leq$5 mg/L). Increases occurred in winter ($\leq$15 mg/L in System 3 and $\leq$17 mg/L in System 4) and summer ($\leq$13 mg/L in System 3 and $\leq$23 mg/L in System 4).

Changes in total Kjeldahl nitrogen result from sedimentation of suspended solids, mineralization of particulate organic matter and marsh plant detritus, immobilization due to assimilation by bacteria and marsh plants, nitrification and volatilization of ammonia. Only volatilization and denitrification, however, provide permanent nitrogen removal mechanisms. Plant uptake can result in permanent nitrogen removal only when the marsh plants are harvested. Denitrification, which involves the transformation of nitrate to gaseous forms of nitrogen by facultative anaerobic bacteria, depends on nitrification for the initial generation of nitrates.

An examination of the TKN concentrations in the channels of System 3 and 4 indicates a relatively gradual reduction between channels (Figure A.20). This suggests that sedimentation plays a minor role in TKN removal. The dramatic suspended solids, BOD and TP decline between the influent and channel B in System 4 was not observed here. Considering that, on average (4 years), ammonia comprises 63% of the TKN in System 3 effluent and 70% of the TKN in System 4 effluent, it appears nitrification plays a major role in TKN reduction. Elevated ammonia levels were associated with low dissolved oxygen concentrations in summer and a combination of low temperatures and reduced oxygen levels in winter. Ammonia removal in summer was enhanced by plant uptake and assimilation by bacteria.

Average total Kjeldahl nitrogen concentrations over the four years were lowest in System 3 (6.1 mg/L), followed by System 1 (7.5 mg/L) and 2 (7.9 mg/L) (Table 5.16). Concentrations in System 4 effluent (8.7 mg/L) were substantially lower than in System 5 (11.1 mg/L). Annual removal efficiencies in System 3 were 32 to 54% and 41 to 55% in System 4 (Table 5.17). As with ammonia removal efficiencies, the lowest treatment efficiencies generally occurred in winter.

5.2.8 Nitrate and nitrite

Monthly average nitrate concentrations in the lagoon effluent were generally below .5 mg/L (maximum 1.6 mg/L). Elevated nitrate levels occurred in some years during the summer or fall, often coinciding with decreases in ammonia concentration. Nitrate levels in the aeration cell effluent were slightly higher with peaks throughout the year reaching a maximum of 3.5 mg/L. Due to the long generation time of autrophic nitrifying bacteria (10-30 hrs.) and the short hydraulic retention within the aeration cell (3 days) very little, if any, nitrification took place during pre-treatment.

Table 5.16 Annual Average Total Kjeldahl Nitrogen Concentration (mg/L) in Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	12.1	6.9	8.2	7.0	21.0	8.2	11.5
1981/82	13.2	9.5	10.6	6.8	18.0	8.6	13.2
1982/83	11.1	6.4	7.1	4.8	17.9	8.0	8.7
1983/84	11.4	7.0	5.5	5.6	17.9	10.1	10.9
4 YR. AVG.	12.0	7.5	7.9	6.1	18.7	8.7	11.1

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5.

Table 5.17 Percent Removal of Total Kjeldahl Nitrogen Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	37.0	20.8	31.6	48.7	38.9
Fall	73.5	73.8	73.6	91.9	89.7
Winter	12.5	6.8	11.5	26.2	23.9
Spring	48.4	22.5	36.7	47.8	39.1
Summer	40.5	25.1	45.7	57.8	30.9
1981-82	19.6	10.2	40.7	46.0	8.7
Fall	-0.8	-28.2	40.7	48.4	-24.9
Winter	15.2	6.4	32.3	27.5	0.1
Spring	26.5	14.9	41.1	47.9	10.8
Summer	32.0	38.6	51.1	59.6	64.0
1982-83	43.1	34.6	54.4	55.4	48.3
Fall	35.0	39.7	55.0	67.0	63.5
Winter	56.4	29.7	49.4	35.3	31.4
Spring	77.0	51.2	75.6	74.4	56.3
Summer	1.4	16.5	35.8	42.9	39.5
1983-84	33.4	48.7	45.6	41.4	35.6
Fall	22.7	58.0	28.5	34.8	18.0
Winter	18.7	11.8	25.1	23.5	13.3
Spring	42.3	37.3	42.9	59.7	47.5
Summer	47.6	81.1	77.6	43.7	42.1
4 Year Avg.	33.3	28.6	43.1	47.9	32.9

Nitrates result from the complete oxidation of organic nitrogen. Nitrification is beneficial in wastewater treatment because it provides a mechanism for ammonia removal and is the preliminary step in permanent nitrogen removal by denitrification. Denitrification takes place under anaerobic conditions by heterotrophic bacteria which use nitrates as an election acceptor when an organic energy source is available. The end products are gaseous forms of nitrogen, such as nitrous oxide and elemental nitrogen gas, and to a lesser extent, nitric oxide. The gases excape into the atmosphere therefore providing the major sink for nitrogen in the marsh environment.

Effluent nitrate levels in all marsh systems were generally less than the detectable limit (<.01 in 1981/82 and <.1 in all other years) or close to it in summer/early fall and winter months (Figure A.21). Increases occurred in the late fall and in spring when sufficient oxygen was available to support nitrification and the production of nitrates. Because bacterial activity and oxygen consumption is highest during summer months, ammonia accumulates until fall when oxygen levels are high enough for nitrification. In fall months, as temperatures progressively decline, the rate of nitrification is reduced. Nitrate levels decline toward the end of winter because the ice cover acts as a barrier to replenishment of oxygen from the atmosphere. In the absence of oxygen, nitrates are utilized as an oxygen source by the marsh bacteria.

These seasonal effects are evident in the loadings and removal efficiencies illustrated in Figure A.22. Treatment efficiencies tended to be highest in summer as a result of the development of anaerobic conditions and uptake by growing plants and bacteria. The channel sampling demonstrated rapid nitrate removal to less than the detectable limit in System 3 in summer (shown as .1 mg/L) and to very low levels in System 4 in all seasons (Figure A.23).

Denitrification takes place in the sediment even though oxygen may be present in the wastewater circulating through the marsh. In submerged soils an anaerobic zone is present below the oxidized sediment surface ("aerobic-anaerobic double layer"). Nitrates formed at the sediment-water interface leach into the anaerobic layer where denitrification takes place.

The microbial reduction of nitrate can follow the assimilatory route where nitrate is reduced to ammonia that is used in bacterial cell synthesis. This provides a mechanism of temporary immobilization although it only occurs when ammonia levels are too low to satisfy the nutrient requirements of the bacteria. Nitrate uptake by marsh plants during the growing season also contributes to the reduction of nitrates in the wastewater.

Average annual nitrate/nitrite concentrations and seasonal and annual removal efficiencies are listed in Tables 5.18 and 5.19, respectively. Nitrate/nitrite concentrations and loadings are difficult to interpret. High nitrate levels represent the first step toward eventual nitrogen removal by denitrification but low levels may indicate that denitrification is keeping pace with nitrate production. Both may be indications that nitrogen is being effectively removed from the system. Effluent nitrate concentration peaks generally occurred when ammonia levels were low and vice versa. The nitrate peaks were of short duration suggesting rapid denitrification following nitrate formation.

5.2.9 Hydrogen Sulfide

Hydrogen sulfide is a soluble, gaseous compound with a characteristic odour which may be detectable at low concentration. It is formed under anaerobic conditions by bacterial reduction of sulfates or decomposition of sulfur-containing organic compounds and is commonly found in domestic and industrial wastewater and in submerged organic soils. Sulfide levels in surface waters are generally low as a result of the transformation of sulfides to sulfates which occur readily in well-aerated waters.

Table 5.18 Annual Average Nitrate/Nitrite Concentration (mg/L) in Marsh Systems

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	.21	.31	.19	.19	.27	.24	.22
1981/82	.17	.24	.17	.17	.38	.16	.22
1982/83	.46	.94	.17	.34	.55	.27	.24
1983/84	.16	.45	.51	.22	.31	.16	.23
4 YR. AVG.	.25	.49	.26	.23	.38	.21	.23

SRLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from systems 1-5.

Table 5.19 Percent Removal of Nitrate/Nitrite Based on Seasonal and Annual Mass Balances

	Systems				
Year/Season	1	2	3	4	5
1980-81	-46.9	1.0	14.2	40.5	33.2
Fall	24.6	22.5	39.2	-149.6	-65.9
Winter	-94.9	-38.2	-57.7	74.4	38.0
Spring	-385.3	-72.4	-11.4	54.0	63.6
Summer	91.3	92.4	93.1	95.5	95.7
1981-82	15.6	50.4	52.8	70.5	59.3
Fall	-57.1	6.4	50.4	76.0	80.1
Winter	-11.2	54.9	33.8	48.1	79.8
Spring	52.3	52.6	66.0	83.4	84.8
Summer	68.6	71.2	74.0	78.1	-54.6
1982-83	-78.7	71.4	36.9	64.5	69.7
Fall	-98.4	61.9	52.5	28.3	-65.8
Winter	-326.5	77.9	-34.9	68.8	76.5
Spring	-207.7	39.7	-44.9	-107.6	21.8
Summer	80.2	80.9	92.8	95.8	95.3
1983-84	-103.2	-126.2	11.8	59.9	38.1
Fall	-122.6	-329.9	-26.2	50.9	52.2
Winter	-255.3	-8.8	-34.0	72.3	81.2
Spring	-84.9	-159.7	44.1	44.4	-40.6
Summer	58.9	51.6	68.2	68.0	63.5
4 Year Avg.	-53.3	-0.9	28.9	58.9	50.1

Aeration cell effluent and lagoon effluent released into the marsh systems contained varying levels of hydrogen sulfide (Figure A.24). Samples were collected primarily in the winter months however some summer data are available. In the lagoon effluent, concentrations above 1 mg/L occurred annually during 2 or 3 winter months indicating reducing conditions under the ice cover in the lagoon. Concentrations in this range in the aeration cell effluent occurred only on one occasion (in Jan. 1982) as generally levels were very low (<0.3 mg/L).

The range of sulfate concentrations were similar in the 2 types of influent (28-250 mg/L), with most samples in the 170-200 mg/L range. Continuous alum additions contributed about 24 mg/L sulfate (50 mg/L of 48% aluminum sulfate continuous feed) to the wastewater (MOE, 1980). The increases from alum probably did not affect sulfide formation as it is more likely that the carbon rather than sulfate was the limiting factor in the biochemical reduction due to the relatively low soluble BOD in the marsh systems.

During the first year of marsh operation when oxygen levels remained high, marsh effluents contained very low levels of hydrogen sulfide. In System 3, concentrations rose above 0.2 mg/L in February only, whereas in System 4 effluent levels were below 0.2 mg/L in all months.

Differences in removal efficiency were evident among the marsh systems. Marshes designed with channels (Systems 1, 3 and 4) performed best as they did not show prolonged winter periods with elevated hydrogen sulfide levels that occurred in systems constructed as shallow marches (Systems 2 and 5). Hydrogen sulfide levels in the effluent of Systems 3 and 4 were low through the winter and early spring (1982/83) or remained low until February or March (1982/82 and 1983/84) at which times concentrations rose to 7.0 and 14.5 mg/L in samples collected from Systems 3 and 4, respectively. In the winter and spring of 1982/83, hydrogen sulfide concentrations in System 3 remained low although the lagoon effluent contained elevated concentrations. Effluent levels in System 3 did not exceed influent levels in winter as was observed in System 4.

The greater hydrogen sulfide production in System 4 in winter is probably related to the higher BOD loadings and greater oxygen depletion in winter observed in this system. The better winter treatment of hydrogen sulfide in Systems 3 and 4 may have been due in part to bubblers placed in the culverts to prevent freezing. In addition to oxidation, the removal of hydrogen sulfide present in wastewater may take place by the formation of insoluble metal precipitates and by the volatilization and loss of hydrogen sulfide to the atmosphere.

Hydrogen sulfide production in summer was observed in all marsh systems in months when sampling was conducted. In Systems 3 and 4, monthly summer levels never rose above 7.5 mg/L, whereas the other systems exceeded this level.

The toxicity of hydrogen sulfide is related to the undissociated molecule, H_2S, rather than its dissociation product (HS^-). The proportion of each compound in solution is established by an equilibrium which is a function of temperature and pH. Unlike ammonia toxicity, hydrogen sulfide toxicity increases as the temperature and pH decreases. For example, at pH 5, it is almost entirely found as H_2S, whereas at pH 7 both species are present in equal amounts and at pH 9 it is almost entirely present in the ionic form (Environment Canada, 1979).

Un-ionized hydrogen sulfide concentrations are shown in Figure A.25. Levels of un-ionized hydrogen sulfide of .002 mg/L or less, the ministry objective for the protection aquatic life in surface waters (MOE, 1984), were only attained in the marsh system effluent at the start of the study (fall, 1980) and in the spring of one or more years in all systems. Annual average hydrogen sulfide and un-ionized hydrogen sulfide concentrations are listed in Table 5.20.

Table 5.20 Annual Average Hydrogen Sulfide Concentration (mg/L) in Marsh Systems

H_2S*

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	1.57	.35	.39	.22	.14	.03	.04
1981/82	2.13	.58	7.70	1.64	.61	2.33	4.06
1982/83	1.11	3.49	6.12	1.11	.10	1.04	3.63
1983/84	2.43	.32	3.54	1.03	.06	1.71	4.66
4 YR. AVG.	1.81	1.19	4.44	1.00	.23	1.28	3.10

Un-ionized H_2S*

YEAR	SBLE	OF1	OF2	OF3	SBAS	OF4	OF5
1980/81	1.20	.19	.23	.11	.07	.01	.02
1981/82	1.40	.27	3.70	.84	.37	1.08	2.39
1982/83	.59	2.95	5.11	.81	.05	.85	2.35
1983/84	2.03	.24	2.93	.85	.04	1.31	3.85
4 YR. AVG.	1.31	.91	2.99	.65	.13	.81	2.15

SBLE = lagoon effluent; SBAS = aeration cell effluent;
OF1-5 = effluent from marsh systems 1-5.

* Analyses performed on different dates in each year, therefore only within-year comparisons are valid.

Based on the Listowel data, large dilutions in summer and winter would be necessary to meet the guideline for un-ionized hydrogen sulfide in the receiving water with continuous discharge from a marsh system. Lower levels of hydrogen sulfide in the marsh effluent may result from an improved pre-treatment (producing an influent with consistently high oxygen content) or post-aeration may be necessary to address this problem.

5.2.1 Phenols

Monthly average phenol levels in the aeration cell effluent and lagoon effluent were generally less than 6.5 ug/L (Figure A.26). Increases in phenol levels to 7-17 ug/L in the lagoon effluent occurred in months during the period of ice cover. Occasional short duration increases in winter ($\leq$10 ug/L) were also observed in the aeration cell effluent. Individual influent samples ranged between <1.-16.5 ug/L.

Phenol concentrations in the marsh effluent samples ranged from <1 to 25 ug/L, with 47 ug/L measured in one sample. The marsh systems receiving lagoon-treated wastewater reduced phenols to low levels in the first winter when oxygen levels were high. In other years large phenol reductions were observed in System 3 until the end of the winter or first month in spring when levels rose in response to the increasing oxygen deficits under the ice. Similar trends were observed in System 4 although influent levels were lower. Systems 2 and 5 showed elevated phenol levels during most winter months.

In summer, phenol levels in the marsh effluents often rose slightly above influent levels. The increase was most pronounced in the channelized systems in the summer of 1983 when severe anoxic conditions were prevalent in the marshes.

Phenolic compounds in domestic sewage, if uncontaminated with agricultural and industrial sources of phenols, are predominantly intermediary products formed by the anaerobic decomposition of organic substances. The reductions in phenol concentrations observed in the marshes in some months can probably be attributed to physical entrapment and chemical adsorption in the soil-vegetation matrix which provide sites for further microbial degradation of these substances. The presence of oxygen reserves in the marsh water prevents the accumulation of phenols and facilitates their breakdown and removal. Increases in phenols in marsh effluents observed during the Study appear to result from the incomplete decomposition of organic substances in the absence of oxygen. In addition, aquatic plants and decaying vegetation are other sources of phenols in the marsh environment.

The ministry guideline for concentrations of phenols in surface waters to protect against tainting of edible fish flesh is that they not exceed 1 ug/L (MOE, 1984). Phenols are toxic at high concentrations (10-100 mg/L). Based on the levels of phenols in the effluent of marsh systems 3 and 4, dilutions in the range of 14:1 would be required to meet the Ministry objective in all months. The recommended pre-treatment, which precludes the disadvantages of pre-treatment methods used at Listowel (i.e. the high phenol concentrations in the lagoon effluent and the buildup of organic sludge from the complete mix aeration cell), may reduce phenols in the effluent. As with hydrogen sulfide, increased dissolved oxygen levels in the marsh would also reduce phenol levels. Post-aeration would lower effluent levels of both wastewater contaminants.

5.3 Bacterial Populations

5.3.1 Sediment Bacteria

The fluctuations of nitrogen and sulphur compounds in wastewater undergoing marsh treatment result in large part from microbial transformations. The bacteria responsible for conversions of wastewater constituents such as nitrifiers, denitrifiers, sulfide reducers and oxidizers, are concentrated in the marsh sediment as the sediment provides attachment sites and large concentrations of bacterial substrates.

The sediment redox potential determines whether bacterial oxidations or reductions predominate in the system. Other factors such as availability of an energy source, temperature, pH, presence of inhibiting substances also influence the diversity and quantities of bacteria in the sediment. Redox potentials in Systems 2, 4 and 5 (Figure 5.13) declined during winter and either increased or declined more gradually prior to steep reductions in summer. The variations in numbers of bacteria in the top 2 cm. of sediment in Systems 2, 4 and 5 during 1981-1982 are illustrated in Figures A.27-A.30. Bacterial levels at the influent and effluent ends of the marsh systems were often quite different reflecting the quality of the wastewater entering the marsh and environmental conditions.

The nitrifiers, Nitrosomonas (ammonia-oxidizers) and Nitrobacter (nitrite-oxidizers), are obligate aerobes. Spring and fall increases in the nitrifier population were coincident with increases in the sediment redox potential. The reduction in nitrifier numbers in summer and winter may be attributed to a combination of reduced oxygen levels and low-winter temperatures which inhibit bacterial metabolism and growth. A two week shutdown of the inflow discharge to all systems during the end of October - beginning of November,

1981, resulted in an increase in nitrifiers, particularly at the influent end of the marsh systems. Nitrosomonas levels were generally higher than Nitrobacter levels. This may result from the fact that Nitrosomonas obtains more energy per mole of nitrogen then Nitrobacter (EPA, 1975). If the cell synthesis per unit energy produced is equal, there should be a greater mass of Nitrosomonas formed than Nitrobacter per mole of nitrogen oxidized.

The growth of the denitrifier population is limited by the presence of oxygen, the lack of a carbon substrate or lack of nitrates. The denitrifiers are comprised of a broad range of heterotrophic bacteria which can utilize nitrate as an electron acceptor in the absence of oxygen. The peaks in denitrifier numbers in April and May, as well as in some summer months, may result from an increased supply of nitrates in the spring and the anaerobic conditions that often develop in summer. Nitrification and denitrification have been shown to occur simultaneously due to the abundance of microsites in the marsh environment.

Sulfate reducers, like the denitrifiers, are anaerobes but sulfate reduction occurs at lower redox potentials. The high sensitivity of these bacteria to oxygen can be seen by their low levels in spring and fall. Sulfide formation was accompanied by increases in the quantity of sulfate reducers. Peak numbers were observed in summer and winter, periods when hydrogen sulfide was detected in the marsh effluents. The highest level was measured in July/82 in System 2 whereas the highest hydrogen sulfide concentration during the two years was measured in Aug./82 in the same system.

5.3.2 Fecal indicators

Fecal coliform (FC) and fecal streptococcus (FS) bacteria are used as indicators of recent fecal contamination of water. When water contains fecal material there is a potential danger that pathogens, or disease causing microorganisms, may he present. However, quantitative relationships between indicators and pathogens are not consistent, one reason being that pathogens are not normal inhabitants of humans or animals.

Levels of FC and FS in the aeration cell effluent and lagoon effluent increased substantially during the winter months. Similar results have heen observed in lagoon treatment systems throughout Ontario. Short-circuiting often occurs in lagoons in winter as a result of density differences which favor surface water movement. Thirty to forty percent of the raw sewage at Listowel comes from a food processing plant which includes poultry processing. As a result the sewage contains high levels of FC, FS and Salmonella. Fluctuations of hacteria in the aeration cell effluent may reflect seasonal changes in the processing plant operation. The drop-off in bacterial loadings in July and August were due at least in part to the mid-July to mid-August shutdown of the plant. The lagoon effluent contained considerably lower levels of fecal bacteria than the aeration cell effluent. For example, in 1982/83, monthly FC and FS geometric mean levels in the lagoon effluent were 0.09-1.9% and 0.04-0.82%, respectively, of the levels in aeration cell effluent prior to release in the marshes.

FC and FS levels in the marshes are shown in Figures A.31 and A.32, respectively. Monthly geometric mean FC and FS densities were generally low in spring and fall (in most cases $\leq$100/100 ml) but rose above 400/100 ml in winter and summer months in most years. Maximum monthly geometric mean FC numbers in Systems 1-5 during the 4 years were 39,000/100 ml, 197,000/100 ml, 8,000/100 ml,

15,000/100 ml and 187,000/100 ml, respectively. In all but System 2, the maximum densities occurred in winter. The FC levels in the effluent of Systems 3 and 4 were similar even though the influent levels in System 3 were 0.1-2.0% of the levels in the influent of System 4. Elevated FC levels in Systems 2 and 5 may have resulted from short-circuiting that was observed in these open shallow marshes. Maximum monthly FS levels in Systems 1-5 were 27,000/100 ml, 557,000/100 ml, 17,000/100 ml, 110,000/100 ml and 723,000/100 ml, respectively. The maximum monthly levels occurred in either July or September.

Reductions in FC and FS densities occurred in the fall, winter and spring in all systems. Treatment efficiencies were reduced during the summer months. In summer, FC and FS levels in the effluent of Systems 1-3 were, in almost all years, greater than densities in the influent which were at their lowest in summer. Although FC and FS levels were elevated in Systems 4 & 5 in summer, because influent bacterial levels were so high in the aeration cell effluent, bacterial reductions remained high throughout the year.

The monthly FC/FS ratio in the lagoon effluent and aeration cell effluent ranged between 2.7-13.7 and 1.1-5.2, respectively, whereas the ratios in the marsh effluents ranged between 0.03-3.8, the lowest ratios (0.03-0.05) appearing in July, August and September. Maximum FC levels were generally several times higher than maximum FS levels in the effluent in winter.

The reduction of fecal bacteria in marsh systems results from natural die-off, exposure to ultraviolet radiation, sedimentation and adsorption onto organic particulates, sediment and plant/litter surfaces. Water temperature affects bacterial levels in the marsh in different ways. Laboratory studies of sewage bacteria indicate that bacteria survival decreases with increasing storage temperature (Berg, 1971). However, there is evidence that elevated temperatures increase the natural predators of bacteria (such as protozoa) and therefore reduce bacterial numbers in the natural environment (Seto, 1984).

The fecal bacteria are facultative anaerobes and therefore are able to survive under low oxygen conditions. The peaks in bacterial numbers in summer and winter coincided with the decline in sediment redox potential during these periods. Other factors related to oxygen deficits which may have had an impact on bacterial survival are increased nutrient availability and the elimination of competition, as only a small fraction of bacteria can survive in the absence of oxygen.

In summer fecal bacteria levels (FC+FS) in systems receiving lagoon effluent exceeded influent levels. This was rarely the case in systems receiving aeration cell effluent due to the very high bacterial levels. The reduction in summer treatment may have been due to accumulations of bacteria which were released from the sediment or attached surfaces during the summer. Fluctuating water levels and the concentrating effect of evapotranspiration could have stimulated this process. The resuspension of fecal indicators in the sediment has been observed in studies of the impact of dredging on water quality (Grimes, 1978). Although bacteriological tracer experiments conducted in System 2 demonstrated a shorter residence time than indicated by dye methods (4 days vs. 6-7 days), the data revealed areas of the marsh where bacteria (antibiotic-resistant E. coli) were retained for more than 30 days (Palmateer et al, 1985). However, fecal bacteria growth may have occurred in the marshes. There is some evidence that growth of enteric bacteria may take place in warm nutrient-rich waters (Seto, 1984). The higher levels of FS as compared with FC in summer may be due to the greater survival ability of FS at high temperatures (Berg, 1971).

Although there were substantial bacterial reductions achieved in the marsh systems, in some months fecal bacterial numbers exceeded the objective of 200/100 ml for disinfected secondary treated effluent. Extended retention in the aeration cell (pre-treatment), however, can be expected to achieve additional bacterial reduction as a result of natural die-off.

5.3.3 Bacterial Pathogens

The detection of four pathogenic bacteria commonly associated with domestic sewage was included in the study. Substantial reductions in pathogen levels were observed in the marsh system effluents. The presence of the pathogens in the lagoon effluent, aeration cell effluent and marsh system effluent is presented in Tables 5.21 - 5.27.

Yersinia enterocolitica is a bacterium which causes gastroenteritis in humans and animals. Y. enterocolitica has the ability to survive at low temperatures during which times reduced competition enhances its growth. High densities of this organism were present in the aeration cell effluent ($\leq$173,000/100 mL) and lagoon effluent ($\leq$152,000/100 ml) during the period of cold water temperatures in winter and early spring. The bacterium was detected rarely in the wastewater released in the marshes during the warmer months.

Systems 3 and 4 demonstrated the best Y. entercolitica removal capacity, followed closely by Systems 1 and 2. In 1981, the bacteria were rarely present in the effluent of Systems 3 and 4. During 1982, when winter influent levels were very high ($\leq$152,000/100 ml in lagoon effluent and $\leq$173,000/100 ml in aeration cell effluent), marsh effluent levels rose to 17,907/100 ml and 4,472/100 ml in System 3 and 4, respectively. During 1983, winter levels were less than 200/100 ml in System 3 and less than 1,000/100 ml in System 4. Y. entercolitica levels in System 5 exceeded 500/100 ml in all three years reaching a maximum density of 50,990/100 ml.

In a survey of the occurrence of Y. entercolitica in the aquatic environment (MOE, 1980), it was found that in surface waters with no apparent contamination from untreated sewage, Y. entercolitica was detected at levels of 2-500/100 ml and was therefore considered the background concentration range. In waters in close proximity to sewage lagoons and in sewage lagoons, levels of 500-30,000/100 ml, and levels up to 100,000/100 ml, respectively, were observed.

Pseudomonas aeruginosa is a bacterium found in fecal wastes and has been identified as the causative agent of a number of infections in humans that may be transmitted through contaminated water bodies. The association between P. aeruginosa and the disease otitis externa, or "swimmers' ear", is of concern in recreational waters.

P. aeruginosa levels in the lagoon effluent (≤390/100 ml) were highest in winter and early spring whereas levels in the aeration cell effluent fluctuated throughout the year (≤1,847/100 ml). P. aeruginosa was detected at low densities (≤10/100 ml) in the effluent of all marsh systems in the fall, spring and summer except in System 5 where spring levels were slightly higher (≤31/100 ml). The levels were elevated in Systems 1, 3 and 4 in winter, remaining less than 60, 67 and 21/100 ml, respectively, but rose above 100/100 ml in Systems 2 & 5 in some winter months. The lowest year-round P. aeruginosa levels were maintained in System 4 although the influent levels were considerably higher than in Systems 1 and 3.

Clostridium perfringens is an anaerobic, spore-forming bacterium which is consistantly associated with human fecal wastes. The bacterium was chosen for monitoring as it is particularly resistant to environmental stresses arising from temperature and pH. Its presence has been demonstrated in chlorinated effluents of secondary sewage plants in the absence of coliforms (Bisson and Cabelli, 1980) and has been suggested as a conservative tracer of fecal pollution.

C. perfringens levels reached a maximum of 4,600/100 ml in the lagoon effluent and 70,000/100 ml in the aeration cell effluent. Effluent concentrations were lowest in the channelized marshes with generally no detectable organisms except during winter and early spring when levels reached 320/100 ml and 580/100 ml in the effluent of Systems 3 and 4, respectively. Levels in effluents of Systems 2 & 5 rose above 1,000/100 ml. The highest levels observed in the lagoon effluent and in all marsh effluents were primarily in winter.

The genus Salmonella contains a large number of species that are pathogenic to man. Because of the established presence of Salmonella bacteria in the raw sewage treated at Listowel, it was not surprising that both the lagoon effluent and the aeration cell effluent almost always contained these bacteria. The marsh systems with channels showed the best removal of Salmonella bacteria. The frequency of occurrence in the effluent of systems receiving lagoon effluent was 42% and 8% in Systems 2 & 3, respectively. System 1 does not represent a valid comparison as tests were conducted there in only 2 months. Systems receiving aeration cell effluent with higher levels (83% vs 67% occurrence in lagoon effluent) achieved reductions to 25% and 73% frequency in Systems 4 and 5, respectively. Of a total of 12 samples of System 3 effluent tested from Jan., 1981 to Oct., 1983, only 1 sample contained detectable levels of Salmonella bacteria.

The better treatment performance of the channelized systems may result from the more controlled path of flow through the channels which increases the retention time by reducing the occurrence of short-circuiting. This provides more bacterial contact with the plants, litter and sediment where bacteria are retained and exposed to ultraviolet radiation and other antagonistic environmental factors that affect bacterial survival. Adequate retention in the marsh is particularly important in winter due to longer survival of pathogenic organisms at low temperatures (Berg, 1971). The recommended 30-day hydraulic detention time in the aeration cell will improve pathogen removal resulting from natural die-off.

Table 5.21 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN LAGOON EFFLUENT (INFLUENT FOR SYSTEMS 1, 2 AND 3).

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	167,715	N	230	8	-
FEB.	48,775	20,000	-	80	110
MAR.	18,330	-	1,270	122	9
APR.	3,158	-	20	20	1
MAY	2,528	N	770	14	-
JUNE	11,733	-	-	27	D
JULY	145	-	0	3	-
AUG.	169	-	-	3	0
SEPT.	449	-	0	9	-
OCT.	629	N	-	8	D
NOV.	-	-	-	-	-
DEC.	2,518	N	-	5	D
JAN. '82	110,000	34,641	1,997	82	-
FEB.	193,649	159,971	4,100	390	D
MAR.	75,423	19,747	1,414	216	-
APR.	7,013	-	4,600	43	N
MAY	98	100	2,500	2	-
JUNE	265	-	2,000	3	0
JULY	120	-	40	1	-
AUG.	262	-	40	3	N
SEPT.	669	-	60	6	-
OCT.	502	N	130	3	-
NOV.	5,523	-	-	22	-
DEC.	5,485	1,000	320	18	D
JAN. '83	4,583	2,828	905	17	-
FEB.	5,959	1,000	900	13	-
MAR.	1,754	448	628	14	-
APR.	539	7,100	1,122	4	-
MAY	454	-	0	8	-
JUNE	206	-	250	2	-
JULY	192	-	10	2	-
AUG.	139	-	20	1	-
SEPT.	98	-	-	2	-
OCT.	553	-	32	8	-
GEOMETRIC MEAN	1,736	4,241	1,036	10	10
FREQUENCY OF OCCURRENCE	100%	63%	88%	100%	67%

Table 5.22 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN AERATION CELL EFFLUENT (INFLUENT FOR SYSTEMS 4 AND 5).

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	442,690	N	6,900	565	-
FEB.	207,123	N	-	379	1,100
MAR.	324,037	20,000	630	806	240
APR.	353,140	-	-	660	240
MAY	374,110	N	5,000	482	-
JUNE	294,161	-	-	925	D
JULY	360,892	-	0	1,036	-
AUG.	39,283	-	-	270	43
SEPT.	220,090	-	5,000	304	-
OCT.	360,153	N	-	428	D
NOV.	-	-	-	-	-
DEC.	390,335	100,000	-	483	D
JAN. '82	448,999	141,421	8,414	273	-
FEB.	641,716	173,205	41,000	1,237	D
MAR.	191,617	49,324	7,133	680	-
APR.	125,064	-	12,100	652	N
MAY	44,186	10,000	25,000	289	-
JUNE	44,523	-	46,000	682	D
JULY	272,198	-	27,000	781	-
AUG.	122,105	-	19,000	986	D
SEPT.	155,350	-	20,000	538	-
OCT.	374,193	N	20,000	1,289	-
NOV.	483,785	-	-	1,358	-
DEC.	351,126	10,000	6,000	1,743	D
JAN. '83	316,228	17,320	7,823	134	-
FEB.	320,312	14,142	16,000	1,249	-
MAR.	151,330	15,874	13,276	1,847	-
APR.	217,686	10,000	5,000	664	-
MAY	236,450	-	14,000	634	-
JUNE	170,137	-	22,000	762	-
JULY	137,757	-	28,000	728	-
AUG.	157,690	-	31,000	1,032	-
SEPT.	262,576	-	-	815	-
OCT.	224,745	-	70,000	515	-
GEOMETRIC MEAN	222,990	28,602	13,148	662	228
FREQUENCY OF OCCURRENCE	100%	69%	96%	100%	83%

Table 5.23 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN SYSTEM 1 EFFLUENT.

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	2,539	1,000	240	1	-
FEB.	987	N	-	6	-
MAR.	28	50	20	1	-
APR.	1	-	-	1	-
MAY	139	N	30	1	-
JUNE	41	-	-	1	-
JULY	192	-	80	1	N
AUG.	362	-	-	1	-
SEPT.	810	-	0	1	-
OCT.	276	N	-	1	-
NOV.	-	-	-	-	-
DEC.	183	1,000	-	1	-
JAN. '82	1,183	1,000	346	3	-
FEB.	38,536	10,488	900	60	-
MAR.	14,480	24,329	393	34	-
APR.	62	-	175	1	-
MAY	3	100	0	1	-
JUNE	99	-	0	1	0
JULY	65	-	0	1	-
AUG.	9	-	0	1	-
SEPT.	10	-	20	1	-
OCT.	2	100	4	1	-
NOV.	1	-	-	1	-
DEC.	3	200	4	1	-
JAN. '83	2	447	5	1	-
FEB.	1	141	12	1	-
MAR.	4	126	21	1	-
APR.	1	700	2	1	-
MAY	4	-	4	1	-
JUNE	516	-	0	1	-
JULY	779	-	0	1	-
AUG.	535	-	0	1	-
SEPT.	938	-	-	1	-
OCT.	24	-	4	1	-
GEOMETRIC MEAN	65	506	10	1	0
FREQUENCY OF OCCURRENCE	100%	81%	68%	100%	0%

Table 5.24 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN SYSTEM 2 EFFLUENT.

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	13,576	-	390	7	-
FEB.	18,879	N	-	153	4
MAR.	260	100	480	7	0
APR.	56	-	-	1	0
MAY	4	N	0	1	-
JUNE	76	-	-	2	N
JULY	1,386	-	0	1	-
AUG.	4,817	-	-	8	1
SEPT.	196,905	-	0	2	-
OCT.	124,794	N	-	2	D
NOV.	-	-	-	-	-
DEC.	1,902	D	-	2	D
JAN. '82	9,222	10,000	303	5	-
FEB.	104,067	33,166	2,000	217	N
MAR.	40,461	12,164	825	103	-
APR.	1,388	-	110	11	N
MAY	80	0	30	1	-
JUNE	484	-	-	1	0
JULY	193	-	0	2	-
AUG.	71	-	0	1	N
SEPT.	44	-	0	1	-
OCT.	60	1,000	0	1	-
NOV.	73	-	-	1	-
DEC.	70	141	0	1	D
JAN. '83	13	141	4	1	-
FEB.	326	141	4	1	-
MAR.	577	464	16	3	-
APR.	2	100	6	1	-
MAY	4	-	0	1	-
JUNE	238	-	10	5	-
JULY	755	-	10	3	-
AUG.	871	-	0	3	-
SEPT.	86	-	-	2	-
OCT.	180	-	0	1	-
GEOMETRIC MEAN	458	247	9	3	1
FREQUENCY OF OCCURRENCE	100%	67%	54%	100%	42%

Table 5.25 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN SYSTEM 3 EFFLUENT.

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	2,113	N	320	4	-
FEB.	188	N	-	3	0
MAR.	1	N	60	1	0
APR.	8	-	-	1	0
MAY	1	N	0	1	-
JUNE	96	-	-	1	N
JULY	307	-	40	1	-
AUG.	471	-	-	1	0
SEPT.	76	-	0	1	-
OCT.	47	N	-	1	N
NOV.	-	-	-	-	-
DEC.	47	3,000	-	1	N
JAN. '82	26	3,162	39	1	-
FEB.	61	1,414	30	1	N
MAR.	7,869	17,907	126	67	-
APR.	278	-	130	3	N
MAY	69	0	100	1	-
JUNE	701	-	0	1	0
JULY	282	-	0	1	-
AUG.	64	-	50	1	N
SEPT.	5	-	0	1	-
OCT.	10	400	0	1	-
NOV.	8	-	-	1	-
DEC.	9	141	0	1	D
JAN. '83	3	173	4	1	-
FEB.	256	100	0	19	-
MAR.	3	215	12	1	-
APR.	1	100	6	1	-
MAY	29	-	0	1	-
JUNE	725	-	0	2	-
JULY	1,755	-	0	6	-
AUG.	394	-	0	1	-
SEPT.	38	-	-	1	-
OCT.	10	-	0	1	-
GEOMETRIC MEAN	53	330	6	1	0
FREQUENCY OF OCCURRENCE	100%	63%	48%	100%	8%

Table 5.26 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN SYSTEM 4 EFFLUENT.

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	15,411	N	580	23	-
FEB.	63	N	-	3	0
MAR.	222	400	450	1	0
APR.	2	-	-	1	0
MAY	210	N	0	3	-
JUNE	500	-	-	1	N
JULY	501	-	0	1	-
AUG.	242	-	-	1	0
SEPT.	215	-	0	1	-
OCT.	292	N	-	2	D
NOV.	-	-	-	-	-
DEC.	33	1,000	-	1	N
JAN. '82	1,251	1,000	124	1	-
FEB.	2,142	4,472	0	3	D
MAR.	3,931	2,884	68	21	-
APR.	49	-	85	1	N
MAY	12	500	0	1	-
JUNE	410	-	-	1	0
JULY	288	-	-	1	-
AUG.	73	-	0	1	N
SEPT.	76	-	0	1	-
OCT.	3	500	0	1	-
NOV.	54	-	-	1	-
DEC.	88	300	0	1	D
JAN. '83	14	316	4	1	-
FEB.	105	316	52	2	-
MAR.	16	262	9	2	-
APR.	1	100	1	1	-
MAY	1	-	0	1	-
JUNE	172	-	0	5	-
JULY	3,036	-	0	10	-
AUG.	573	-	0	3	-
SEPT.	1,120	-	-	1	-
OCT.	293	-	0	1	-
GEOMETRIC MEAN	121	563	4	2	0
FREQUENCY OF OCCURRENCE	100%	75%	39%	100%	25%

Table 5.27 LEVELS OF FECAL COLIFORMS (FC), YERSINIA ENTERCOLITICA (YE), CLOSTRIDIUM PERFRINGENS (CP), PSEUDOMONAS AERUGINOSA (PA), SALMONELLA SPECIES (S SP) IN SYSTEM 5 EFFLUENT.

DETECTED (D), NOT DETECTED (N), NOT CONDUCTED (-)

SAMPLING PERIOD	BACTERIAL LEVELS (COUNT/100 ml)				
	FC	YE	CP	PA	S SP
JAN. '81	99,139	42,426	1,300	218	-
FEB.	9,121	5,477	-	33	9
MAR.	45,111	1,000	380	97	4
APR.	71	-	-	1	D
MAY	725	N	0	31	-
JUNE	693	-	-	4	D
JULY	12,488	-	0	14	-
AUG.	16,313	-	-	2	1
SEPT.	90,882	-	0	6	-
OCT.	51,717	N	-	25	D
NOV.	-	-	-	-	-
DEC.	6,560	2,000	-	8	D
JAN. '82	152,840	24,495	1,442	61	-
FEB.	187,029	50,990	1,400	482	N
MAR.	29,350	11,402	548	138	-
APR.	552	-	140	4	N
MAY	10	200	-	1	-
JUNE	*	*	*	*	*
JULY	*	*	*	*	*
AUG.	54	-	200	1	D
SEPT.	161	-	60	1	-
OCT.	83	2,500	4	3	-
NOV.	509	-	-	3	-
DEC.	6,973	447	0	53	D
JAN. '83	5,683	141	42	37	-
FEB.	1,414	548	70	12	-
MAR.	8,713	632	18	53	-
APR.	24	1,000	16	1	-
MAY	198	-	4	1	-
JUNE	246	-	0	2	-
JULY	1,179	-	0	6	-
AUG.	1,295	-	0	4	-
SEPT.	1,003	-	-	6	-
OCT.	15,142	-	0	2	-
GEOMETRIC MEAN	2,317	2,224	19	9	2
FREQUENCY OF OCCURRENCE	100%	88%	64%	100%	73%

*System drained

5.4 Plant Biomass and Harvesting

5.4.1 Nutrient removal by harvesting

The cattail harvest in the marsh systems represented nitrogen and phosphorus removals of 1.2% to 9.6% of the total nutrient input from wastewater plus precipitation (Tables 5.28 and 5.29). During 1981, the first year of marsh operation, maximum N and P removal was 9.1% and 9.6%, respectively, whereas the percentage declined to 2.4% and 1.2%, respectively, in 1982. This decline was due to the higher plant biomass production in the first year. Nutrient loadings during this period were double loadings in succeeding years. Other factors, such as the newly submerged soil and the recent propagation of the cattails from rhizomes, may have played a role in the production of the high biomass in the first year. The dry weight of cattails harvested in System 4 was 5 to 6 times higher in 1981 than in 1982, the 2 years when single harvests precluded regrowth as a factor.

The nutrient levels in the harvested cattails are listed in Table 5.30. The average concentration of nitrogen in harvested cattail shoots was 27.7 mg/g (range 16.5-52.6 mg/g), whereas the average phosphorus concentration was 2.4 mg/g (range 1.2-3.8 mg/g). In 1981, nitrogen and phosphorus levels in the first harvest of cattails in Systems 3 and 4 were higher in July than in August. Other researchers have observed a similar decline in nutrient levels in above-ground cattail structures as the growing season progresses from a maximum early in the season (Klopatek, 1978). The high nutrient content of the cattail shoot material harvested in September, 1983 (the third harvest of the season) suggests a large proportion of succulent, new growth. In 1981 when both Systems 3 and 4 were harvested, nitrogen and phosphorus levels in the July harvest were higher in System 4 than in System 3. This difference may have resulted from the higher nutrient loadings in System 4.

Table 5.28 Total Phosphorus Removed By Cattail Harvest

Year	Date	System	Channels	Total Ann. TP Input to System (g/m^2)	Total Ann. TP Output from System (g/m^2)	Total Ann. % Removal Efficiency	Dry Wt. Harvested (g/m^2)	TP in Harvested Material (g/m^2)	% of Total Ann. TP Input Removed by Harvest
1981	July 16 Aug. 26	III	4 & 5 1,2 & 3	14.4	6.4	55.3%	534.6	1.39	9.6%
1981	July 16 Aug. 26	IV	4 & 5 1,2 & 3	33.1	6.4	80.8%	591.6	1.67	5.1%
1982	Aug. 24	IV	1-5	15.3	2.56	83.3%	94.6	0.18	1.2%
1983	July 7 Aug. 18 Sept. 29	IV	2 & 4 2 & 4 2 & 4	16.3	1.88	88.5%	35.8 62.5 60.6	0.09 0.09 0.18	
							T = 158.9*	T = 0.36*	2.3%
1984	July 11 Aug. 9 Sept.	IV	2 & 4 2 & 4 2 & 4	20.6	3.76	81.8%	91.3 111.2 126.5	0.21 0.27 0.31**	
							T = 329.0*	T = 0.79*	3.8%

* The average dry wt. and nutrient level harvested in channels 2 & 4 were applied to each of 5 channels.

** Nutrient levels in Sept. samples were not available therefore August data were used to calculate this value.

Table 5.29 Total Kjeldahl Nitrogen Removed By Cattail Harvest

Year	Date	System	Channels	Total Ann. TKN Input to System (g/m^2)	Total Ann. TKN Output from System (g/m^2)	Total Ann. % Removal Efficiency	Dry Wt. Harvested (g/m^2)	TKN in Harvested Material (g/m^2)	% of Total Ann. TKN Input Removed by Harvest
1981	July 16 Aug. 26	III	4 & 5 1,2 & 3	156.0	98.3	37.0%	534.6	14.20	9.1%
1981	July 16 Aug. 26	IV	4 & 5 1,2 & 3	199.0	90.6	54.5%	591.6	16.58	8.3%
1982	Aug. 24	IV	1-5	89.0	32.2	63.8%	94.6	2.19	2.4%
1983	July 7 Aug. 18 Sept. 29	IV	2 & 4 2 & 4 2 & 4	99.6	32.7	67.2%	35.8 62.5 60.6	0.96 1.12 2.16	
							T = 158.9*	T = 4.25*	4.3%
1984	July 11 Aug. 9 Sept.	IV	2 & 4 2 & 4 2 & 4	131.9	64.7	51.0%	91.3 111.2 126.5	2.23 2.45 2.78**	
							T = 329.0*	T = 7.46*	5.7%

* The average dry wt. and nutrient level harvested in channels 2 & 4 were applied to each of 5 channels.

** Nutrient levels in Sept. samples were not available therefore August data were used to calculate this value.

Table 5.30 Nutrients in Harvested Cattail Shoots

				TKN (mg/g)			TP (mg/g)		
Year	Date	System	Channels	Mean	Range	% of Dry Wt.	Mean	Range	% of Dry Wt.
1981	July 16	III	4 & 5	39.1	(38.5-39.6)	4.0%	3.1	(2.8-3.4)	.31%
	Aug.26	III	1,2 & 3	22.6	(21.2-23.0)	2.3%	2.4	(2.4-2.5)	.24%
1981	July 16	IV	4 & 5	49.1	(45.5-52.6)	4.9%	3.9	(3.8-4.0)	.39%
	Aug.26	IV	1,2 & 3	22.1	(18.5-27.3)	2.2%	2.2	(1.6-2.7)	.22%
1982	Aug.24	I	4 & 5	23.3	(22.0-24.5)	2.3%	2.1	(2.0-2.2)	.21%
1982	Aug.24	IV	4 & 5	23.8	(21.4-26.2)	2.4%	1.7	(1.2-2.2)	.17%
	Aug.24	IV	1,2 & 3	24.1	(19.1-28.5)	2.4%	2.1	(1.9-2.3)	.21%
1983	July 7	IV	2 & 4	26.9	(26.3-27.5)	2.7%	2.6	(2.5-2.6)	.26%
	Aug.18	IV	2 & 4*	18.1	(16.5-19.6	1.8%	1.5	(1.3-1.6)	.15%
	Sept.29	IV	2 & 4*	35.4	(32.3-38.5)	3.5%	2.9	(2.7-3.1)	.29%
1984	July 11	IV	2 & 4	24.4	(23.3-25.5)	2.4%	2.3	(2.2-2.3)	.23%
	Aug. 9	IV	2 & 4*	22.0	(21.0-22.9)	2.2%	2.5	(2.4-2.5)	.25%
	Sept.	IV	2 & 4*	NA	NA	NA	NA	NA	NA

NA=data not available
* regrowth following previous harvest.

6868g

Nutrient removal was increased with repeated cuttings. In the 2 channels of System 4 that received a single (August) cutting in 1982 and three cuttings (in July, August and September) in 1983, although nutrient inputs were similar in the 2 years, there was a doubling in nutrient removal from the multiple harvest. Plants harvested at the fruiting stage did not show regrowth following the first cutting in mid-July. Less mature plants grew up to 81% of their initial height in 7 weeks after cutting.

These data indicate that even with multiple harvesting of cattails from the artificial marsh systems, at the loadings experienced in 1982-1984, harvesting removed a small part (<6%) of the total annual nutrient input. A considerable amount of nutrients in cattail plants are stored in the underground parts, roots and rhizomes, particularly in the latter part of the growing season when there is a net movement of nutrients from the shoots into the winter storage organs. The rhizomes of whole plants removed from the artificial marsh in the month of August represented 42% of the dry weight of the whole plant (Pos, 1983). The harvesting apparatus left approximately 20% of the shoot material and all underground parts unharvested. Therefore, only 46% of the cattail plant was harvested.

It appears that the major role of marsh plants in wastewater treatment is the favourable environment they provide for treatment mechanisms which depend upon bacterial metabolism, adsorption and precipitation reactions. However, nutrient assimilation by marsh plants provides some immobilization of nitrogen and phosphorus during the summer period when receiving waters are most sensitive to nutrient input. Harvesting may reduce the possible long-term problems in marsh operation associated with the accumulation of organic matter. Until this is shown to be necessary, the costs associated with harvesting including labour and land requirements for broad earth berms (to support the harvesting equipment) do not appear to be justified by the benefits.

5.4.2 Resources Recovery

A byproduct of artificial marsh sewage treatment is the production of plant biomass which may be harvested to yield useable products. Harvested plant biomass could be utilized for livestock feed, compost and soil amendments or converted into energy.

Preliminary studies on the generation of methane from freshly harvested cattails were conducted at the University of Guelph, with funds provided by the Ministry of Energy. Biogas production, $1.1m^3/m^3$, compared favourably with more conventional digester feedstocks, such as, sewage sludge and animal manures which yield $1\text{-}2m^3/m^3$. The yield from cattails would be enhanced by harvesting more of the lower portion of the plants which contains a major part of the digestible protein in the plant.

Of greater significance, however, may be the use of digester solids as a protein supplement in livestock feed. The solid digester residue represents more than 60% of the dry influent feedstock (cattails) and is higher in protein than the unprocessed plant material. It would have to be determined if the digested feedstock is palatable and digestible for animals.

The data indicate the feasibility of uses for harvested cattails, however, the economic viability of resources recovery would have to be investigated. The economics of resource recovery will depend on the availability of local markets, final uses and values of the products, and harvesting and processing costs.

5.5 Litter component

The cattail litter decomposition experiment was conducted over a 2 year period at a natural marsh in the Town of Bradford. Although the marsh did not receive wastewater before or during this study, the results provide information on the fate of unharvested plant biomass and nutrient cycling within a marsh. However, some differences in decay rates may arise in marshes used for wastewater treatment as a result of 1) higher nutrient levels in the plant litter and 2) variations in environmental factors (e.g. temperature and aeration). The suspended and submerged litter bags filled with cattail shoot material were subjected to environmental conditions generally similar to the exposure of naturally fallen litter. The exception is the exclusion of invertebrates larger than the mesh-bag openings and the prevention of fragmentation in the suspended litter.

The process of decomposition of cattail litter begins in the aerial environment after the end of the growing season. The standing litter is reduced in weight and nutrients by leaching and fragmentation from grazing and the impact of weather. The fallen shoots form mounds partially exposed to the air or become submerged in the marsh water. Of particular interest in the context of marsh wastewater renovation is 1) the extent and seasonality of nutrient transfer (release and immobilization) between decomposing litter and the circulating water, and 2) the rate of loss of dead plant material.

The submerged Typha litter samples retained, on average, 65% of their dry weight at the end of the first year and 55% at the end of the 2 year period (Figure 5.15). A 10% reduction in weight took place during the first 2 weeks. Other researchers have observed this type of dramatic weight loss and suggested it was due to leaching of soluble compounds (Peterson and Cummins, 1974). During senescense cell membrances lose their integrity causing a release of

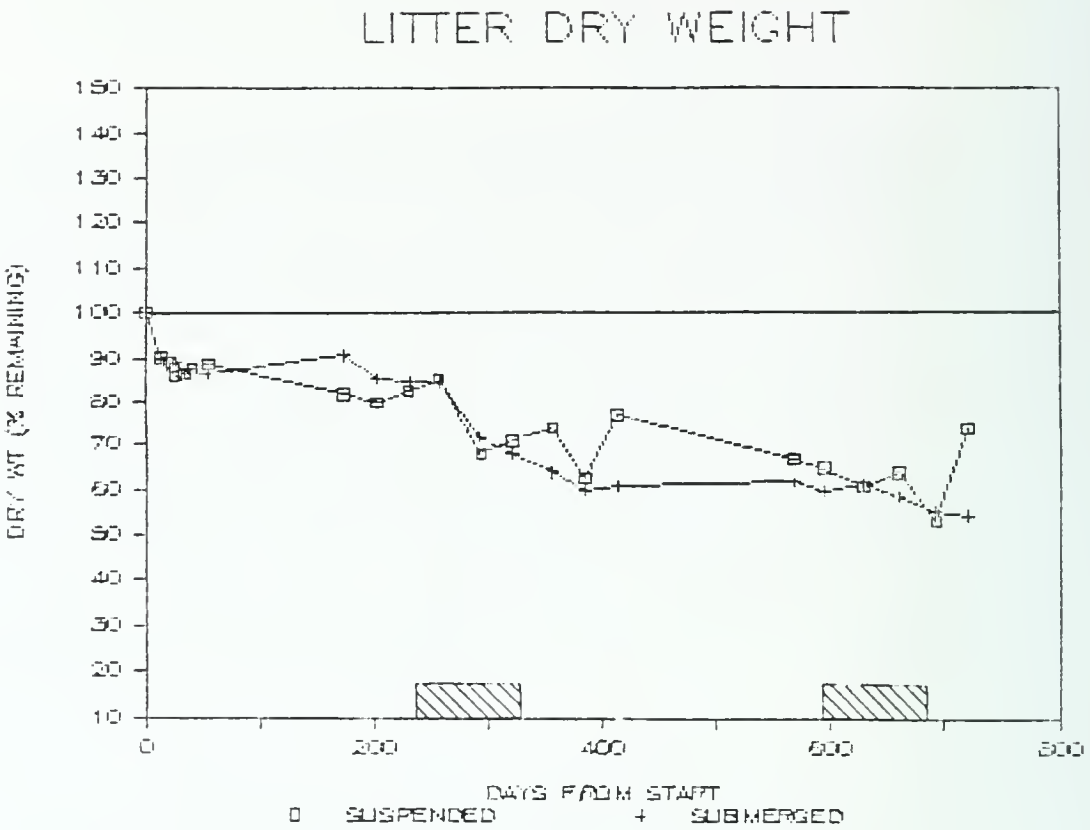


Figure 5.15 Dry weight loss in suspended and submerged Typha litter from start of experiment (Oct. 18, 1979) to end of 2 year period. (Hatching indicates summer periods, June-August).

dissolved materials. The steepest decline occurred during the first summer and early fall when the samples lost 25% of their weight. This period coincides with higher water temperatures and the resultant increased microbial activity. A very slight reduction took place during the following summer for, although temperatures were favourable for microbial growth and decomposition, the remaining plant material contained a smaller proportion of easily oxidized energy sources (e.g. simple sugars and amino acids) than during the first summer.

The suspended litter demonstrated a similar decline in weight, with a somewhat greater weight retention then submerged litter during the second year of the study. The more irratic decline in weight may reflect the timing and severity of the weather. The greater weight loss of the submerged litter during the second year may result from better access to decomposer bacteria and invertebrates, the presence of continuous moisture required for the growth of bacteria and fungi and the greater availability of dissolved nutrients in the aquatic environment.

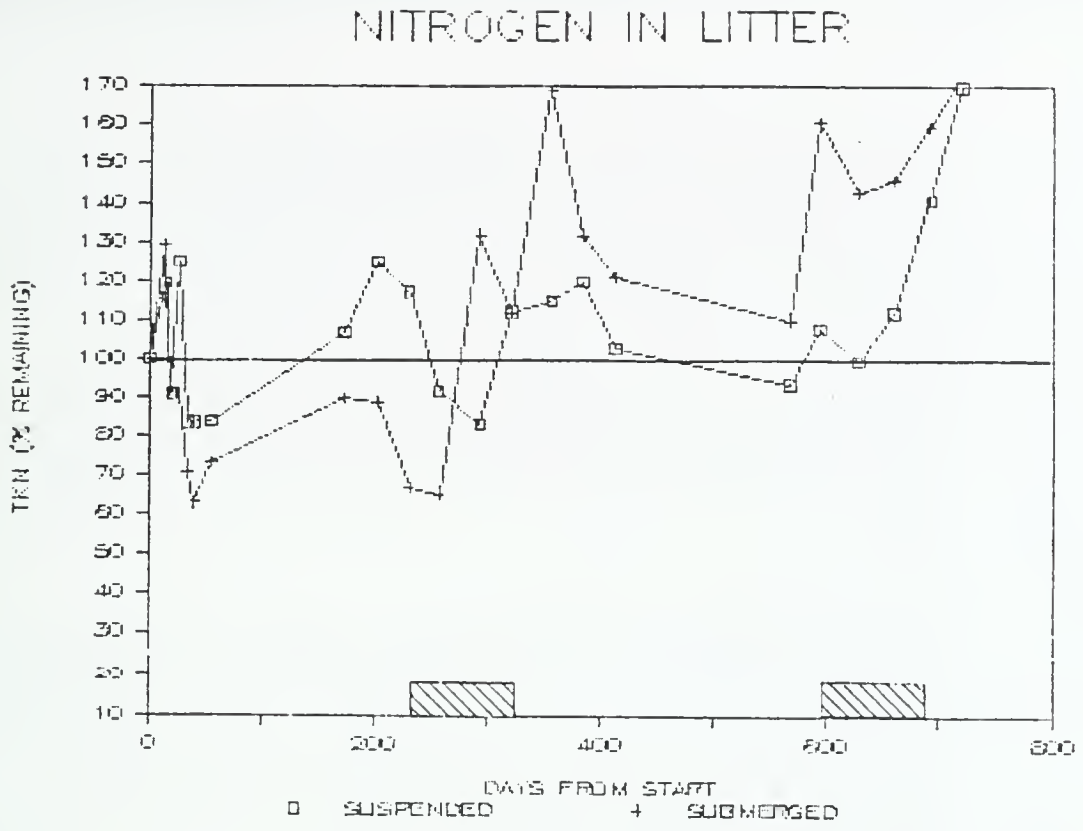
Based on observations of several researchers that litter decomposition follows a negative exponential curve, a weight loss parameter calculated from weight loss at one year has been used to extrapolate turnover rates of litter in wetlands (Chamie and Richardson, 1978). The weight loss paramater, k, was calculated from the fractional weight loss, k', in the equation $k = -\ln (1-k')$. Using this method, the projection of the half-life, 95%-life and 99%-life for the submerged cattail litter at Bradford was 1.5, 6.7 and 11.1 years, respectively.

High decay rates in aquatic plant litter has been associated with low total fiber (hemicellulose, cellulose and lignin), high initial nitrogen content and low initial C:N ratio (Godshalk and Wetzel, 1978). This work with regard to the interaction of temperature and aeration on rates of litter decomposition yielded a gradient from very low under cold, anaerobic conditions to rapid under warm, aerobic conditions.

The nutrient data show a net accumulation of N and P in the submerged litter samples at the end of the 2 year period (Figure 5.16). The immediate increase in litter nutrient levels followed by a steep decline, particularly severe for P, indicates leaching of nutrients. Most accumulation of nutrients occurred during the growing season, beginning after the start of summer in the first year and in late spring in the second year. This suggests that as microbial activity increased with increases in temperature, nutrient levels in the litter were insufficient. Researchers have demonstrated nitrogen uptake from water by microorganisms growing on nutrient deficient submerged aquatic plant litter (Almazon and Boyd, 1978).

The maximum % N and P remaining in the submerged litter compared with the initial level was 173% and 133%, respectively, whereas the minimum % N and P remaining was 65% and 35%, respectively. Nitrogen was accumulated to a greater extent than phosphorus in relation to the initial level, as well as in absolute terms, as the initial N:P ratio in the litter ranged between 7:1-17:1. Nutrient releases occurred intermittently but primarily in the fall.

Following the initial P loss from the submerged litter it was not until the end of the 2 years before there was substantial P immobilization in the litter. In contrast, the immobilization of N from the wastewater began in mid-summer of the first year (275 days) after which net accumulation of N was maintained until the end of the experiment. Less accumulation of nutrients was observed in the suspended litter. This was probably due in large part to the lower microbial population associated with the suspended plant material and infrequent contact with the wastewater. Presumably a substantial portion of nutrient accumulation in the suspended litter originated from atmospheric deposition.



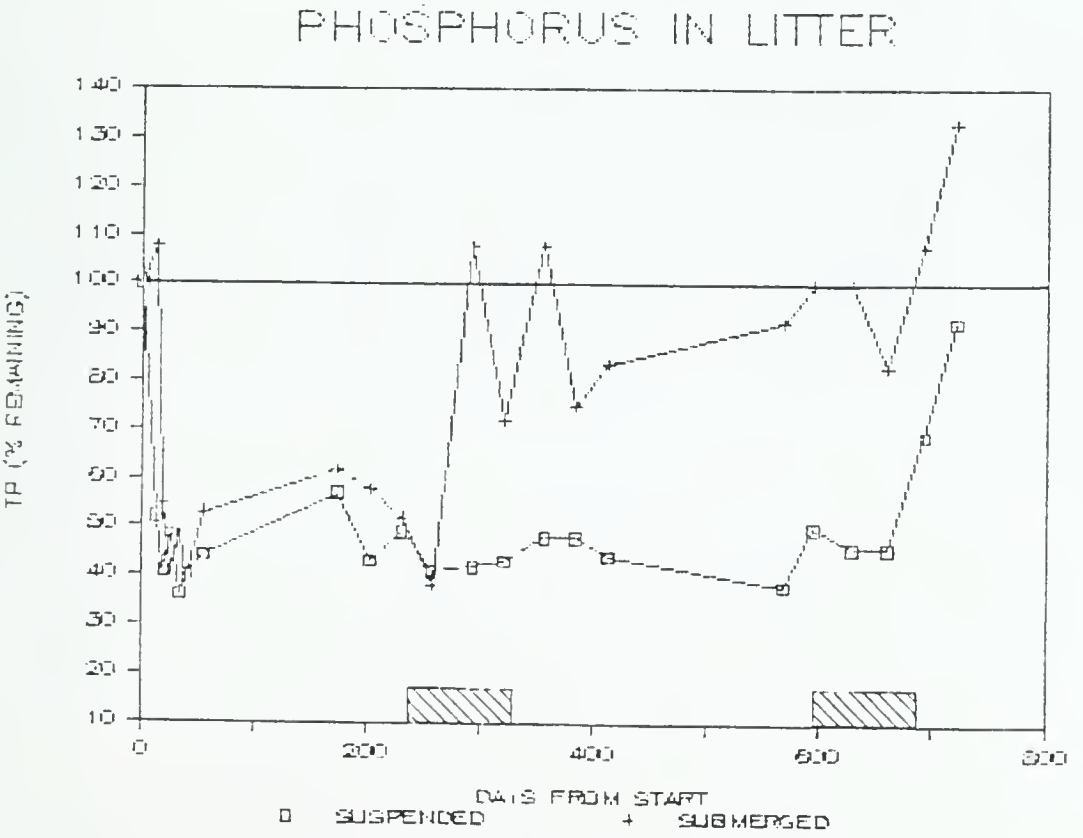


Figure 5.16 Total Kjeldahl nitrogen and total phosphorus concentrations in suspended and submerged litter from start of experiment to end of 2 year period. (Hatching indicates summer periods, June-August)

These data indicate that natural litter removal in the marsh is a relatively slow process. A part of the plant material produced in each growing season will remain in the marsh for a number of years. This may be an advantage as several wastewater renovation processes, such as assimilation by bacteria attached to the litter and all hetrotrophic microbial transformations, are dependent on a substantial organic component in the marsh. Although interference with marsh operation resulting from excessive litter accumulations may occur in the long-term, there was no evidence of this during the study.

At Bradford, mild increases in litter nutrient levels in winter may indicate surface adsorption as microbial activity is inhibited at low temperatures. Nutrient immobilization, although temporary, took place primarily in summer, a period when evapotranspiration increases the level of soluble nutrients and receiving waters are usually most sensitive to effluent release.

5.6 Nuisance Organisms

Mosquitoes and midges are the primary organisms of concern in marshland treatment of wastewaters. Studies at Listowel revealed that the numbers of midges generated from the marshes were small in comparison to the production of these organisms in conventional lagoons. On the other hand, vast numbers of mosquitoes were produced in the marsh environment, reaching peak populations of 9,000 larvae per m^2. Four species of mosquitoes were identified, however, none are significant man-feeders.

The man-feeding Mansonia perturbans which typically occurs in wetlands was not found at the Listowel site. Adult trapping data showed that mosquitoes produced in the Listowel marshes were not contributing to increases in populations beyond a 1 km radius. The only mosquito species of concern is Culex pipiens, a probable vector of St. Louis Encephalitis. Consequently, chemical control would only be required in the event of encephalitis activity. A recently registered bacterial insecticide, Bacillus thuringiensis var. israeliensis which provided 90% control in trials at Listowel, is recommended for this purpose.

The mosquito populations can be regulated to a large extent by management of vegetation on the berms. In Listowel, population densities of Culex pipiens were inversely related to surface water coverage by duckweed (Lemna spp.) growth. Observations indicated that small pockets of clear water serve as the primary egg laying sites for Culex spp. Removal of the berm vegetation which shades the water surface should provide conditions for a more complete duckweed cover and thereby significantly reduce the number of egg laying sites.

6.0 CONCLUSIONS AND RECOMMENDATIONS

6.1 Wastewater Renovation in the Artificial Marshes

Marsh treatment of wastewater is accomplished by physical, chemical and biological processes which include sedimentation, filtration, precipitation, adsorption, decomposition, bacterial and plant metabolism and assimilation and natural die-off. Wastewater retention time was found to be an important factor in marsh wastewater treatment and this is reflected in the recommendations for marsh design and operation. Marsh operating procedures included the regulation of marsh hydrology to compensate for seasonal influence on the rate of flow, and thus, retention time. The water depth was adjusted seasonally to maintain an appropriate retention time for optimum contaminant removal. The objective was to obtain contaminant removal at a level that would produce an effluent quality suitable for continuous year-round discharge. The Listowel data suggest that at a given contaminant loading and retention time, seasonal influences relating to insufficient dissolved oxygen concentrations and secondarily, low water temperatures, were responsible for most of the variability in contaminant removal.

Influent and effluent concentrations of major wastewater constituents plus removal efficiencies based on annual loadings in Systems 3 and 4 are summarized for the four year period in Table 6.1. The comments on the capabilities of marsh treatment demonstrated at Listowel refer mostly to System 3. The reason for this is the superior performance of the channelized design and the similiarity of the System 3 (lagoon pre-treated) influent with levels of wastewater constituents (e.g., SS, BOD_5 and TP) which would be achieved with the recommended pre-treatment.

Table 6.1 Average Wastewater Concentration and Treatment Efficiency in Systems 3 and 4 Over Four Years

SYSTEM 3: CHANNELIZED MARSH RECEIVING LAGOON EFFLUENT

	4 Year Average Concentration (mg/L)		% Removal Based on Mass Balance	
	Influent	Effluent	4 yr. Avg.	Ann. Range
Suspended Solids	22.8	9.2	60.5	(40 to 75)
BOD-5	19.6	7.6	59.4	(56 to 63)
Total Phosphorus	1.0	0.5	46.0	(39 to 53)
Soluble Reactive Phosphorus	0.37	0.26	9.3	(-12 to 59)
Total Kjeldahl Nitrogen	12.0	6.1	43.1	(32 to 54)
Nitrate-Nitrite	0.25	0.23	28.9	(12 to 53)
Ammonia	7.2	3.8	38.8	(31 to 51)
Un-Dissociated NH3	0.13	0.01		
Hydrogen Sulfide	1.8	1.0		
Un-Dissociated H2S	1.31	0.65		

SYSTEM 4: CHANNELIZED MARSH RECEIVING AERATION CELL EFFLUENT

	4 Year Average Concentration (mg/L)		% Removal Based on Mass Balance	
	Influent	Effluent	4 yr. Avg.	Ann. Range
Suspended Solids	111.1	8.0	92.6	(90 to 94)
BOD-5	56.3	9.6	81.6	(78 to 88)
Total Phosphorus	3.2	0.6	79.1	(76 to 84)
Soluble Reactive Phosphorus	0.40	0.34	-24.8	(-138 to 59)
Total Kjeldahl Nitrogen	18.7	8.7	47.9	(41 to 55)
Nitrate-Nitrite	0.38	0.21	58.9	(41 to 71)
Ammonia	8.6	6.1	22.5	(5 to 35)
Un-Dissociated NH3	0.07	0.02		
Hydrogen Sulfide	0.2	1.3		
Un-Dissociated H2S	0.13	0.81		

Large reductions in suspended solids and BOD_5 were achieved on a year-round basis. Removal efficiencies were generally lowest in spring or summer as phytoplankton densities and their contribution to the organic content of the suspended particles were highest at this time of year. Average concentrations of both constituents in each year were less than 10 mg/L in System 3, well below the current design guidelines for conventional activated sludge secondary plants.

Phosphorus removal was greatly facilitated by chemical pre-treatment. Phosphorus levels were reduced further in the marsh by adsorption in the litter-sediment complex and assimilation by aquatic flora. Total phosphorus was reduced to annual levels averaging 0.5 mg/L in System 3, below the 1 mg/L objective for conventional processes with phosphorus removal.

The data suggest a reduction in the soluble phosphorus retention capacity in all Listowel marsh systems during the four years. The phosphorus retention capacity of the sediments appeared to be related to cumulative phosphorus loadings. The decline in annual soluble phosphorus removal was particularly pronounced in systems receiving high phosphorus loadings from the aerated effluent (e.g. System 4). The high phosphorus loadings in all systems during the first year of the study probably hastened the soluble phosphorus saturation of the sediment. In addition, periodic oxygen depletion in the marshes (and low sediment redox potential) has been associated with increased solubilization of phosphorus and may have contributed to reduced phosphorus retention.

As evidence suggests that the sediment mineral content and structure influences the phosphorus adsorption capacity, running silt-laden water (e.g. stormwater) through the marsh may extend this capacity by increasing the sites for adsorption and coprecipitation of phosphorus. The maintenance of a highly oxygenated influent, as would be accomplished by the recommended pre-treatment, may also improve sediment phosphorus retention. The lack of a permanent phosphorus sink in the marsh may necessitate the periodic recharge of the phosphorus retention capacity of the sediment (e.g. dredging).

Elevated ammonia levels in winter and in summer were observed in the effluent of the Listowel marsh systems. These levels could pose a potential danger to aquatic life in a continuous release system if the receiving water did not provide an adequate dilution to reduce the oxygen demand imposed by elevated ammonia levels and to meet Ministry objectives for un-ionized ammonia. Based on the Listowel results, the ammonia criterion for release into surface waters would limit the application of marsh treatment to sites with the necessary dilution.

The data suggest that nitrification, the principal ammonia removal process, was inhibited in summer by limitations imposed by an inadequate oxygen supply whereas in winter, alternatively or in combination, low oxygen levels and low water temperatures were limiting factors. A 10:1 dilution would be required in summer and 5:1 dilution in winter (based on the maximum monthly average ammonia level in System 3 effluent in each season), to meet the un-ionized ammonia objective, assuming an elevated pH of 8 in the receiving waters. These dilutions, after mixing, would reduce maximum monthly average total ammonia concentrations to <1 mg/L and 2.6 mg/L in summer and winter, respectively. However, greater reductions in ammonia, as well as hydrogen sulfide and phenols, are expected with extended aeration pre-treatment as oxygen levels entering the marsh will be maintained at a consistently high level.

Elevated hydrogen sulfide levels were detected in the marsh effluent toward the end of the winter and summer when anaerobic conditions were most severe so that oxygen and nitrates were unavailable to satisfy the requirements of bacterial metabolism. The dilution required to meet ministry objectives of .002 mg/L un-ionized hydrogen sulfide could rarely be met. Hydrogen sulfide levels can be lowered to acceptable levels with the addition of post-aeration treatment. Experience with lagoons indicates that a post-aeration cell with 1-day retention would control hydrogen sulfide in the marsh effluent while requiring small amounts of energy to operate. In the presence of oxygen, sulfides are readily oxidized to sulfates.

Phenol levels were reduced to less than the detectable limit during the first year when oxygen levels remained high but, in following years, were often elevated toward the end of summer and/or winter, on some occasions exceeding influent levels. Increases in phenols in the marsh effluents appeared to be associated with low oxygen conditions. Dilutions up to 14:1 would be required to meet ministry objectives of 1 ug/L in surface waters. Because phenol production is a result of incomplete decomposition of organic compounds, post-aeration would also significantly reduce phenol concentrations in the marsh effluent. However, the recommended extended aeration pre-treatment may sustain sufficient oxygen levels in the marsh to reduce phenol and hydrogen sulfide levels in the marsh effluent.

Fecal bacteria reductions were high during most of the year with effluent fecal coliform levels less than or equal to those of disinfected secondary effluent (200/100 ml). However, elevated monthly geometric mean numbers (maximum 8,000/100 ml in System 3) occurred in summer and/or winter. Bacterial pathogens were generally detected in the effluent at low levels except for increases in some winter months. Salmonella was only detected in System 3 in one out of twelve samples collected over the four years. Longer retention during pre-treatment in the aeration cell recommended in this report will provide greater opportunity for natural die-off.

6.2 Design and Management Recommendations

6.2.1 Site Selection and Land Requirements

Artificial marshes may be constructed anywhere subject to the availability of land and proximity to a wastewater source. Since grading and excavating represent a major cost factor, a low relief topography and relatively impermeable soils, such as the clay soils at Listowel, are preferable. Alternatively, sealers can be used to prevent infiltration and exfiltration in the marsh basin. In areas with heavy clay soils, top soil or organic amendments are necessary to provide a growth medium for the rooted emergent vegetation. An even soil surface reduces the potential for short-circuiting, however, a slight grade is recommended so that the marsh can be drained, if necessary.

Land requirements for marsh wastewater treatment facilities depend on hydraulic loadings. At a loading rate of 200 m^3/ha/d, a 1 migd (4,546 m^3/d) community will require 23 ha (55 acres) of marsh surface plus additional land for internal flow regulation and pre-treatment facilities. This compares favourably with the conventional seasonal lagoon with 6 months (180 days) storage capacity which requires 55 ha (135 acres) for an equivalent sewage loading (assuming 1.5 m depth).

6.2.2 Vegetation

Plants play an important role in marsh wastewater treatment. Their contribution to wastewater renovation is achieved by the assimilation of nutrients, the buildup of organic matter (i.e. plant detritus) in the sediment, the provision of adsorption sites and attachment surfaces for bacteria and other biota, the regulation of the water flow, filtration and sedimentation.

Cattails (Typha spp.) are among the most common marsh plants in Ontario. They are aggressive perennials that form large clones by vegetative growth and demonstrate a high percentage survival following the establishment of permanently flooded conditions. Cattails are easily propagated from rhizomes collected from any poorly drained area and produce a large annual biomass. From the experience at Listowel, cattail rhizomes planted at 1 m intervals in a suitable soil can be expected to grow into a dense stand within 3 months in southern and central Ontario. Although the cattail biomass was higher in the first year than in succeeding years, a complete cover crop of cattails was maintained throughout the study. The reduction in biomass (50-80%) in years 2 through 4 compared to the first year, may have been related to the reduction in annual nutrient loading (40-50%) during this period or other factors associated with recent marsh construction, soil submersion and plant propagation.

Harvested cattails removed less than 10% of the total annual nitrogen and phosphorus input to the marsh systems during the four years. Only 45% of the dry weight of the plant is actually harvested as 20% of shoot material is below the water and rhizomes are rich in proteins and nutrients. Although multiple harvests increased the nutrient removal by 2-3 times, at nutrient input levels maintained during the last 3 years, the percentage of total input removed remained low (<6%).

The data suggest that the practice of harvesting for nutrient removal is not cost-effective when factors such as labour, energy and the additional land required for wide berms are taken into consideration. The long-term need for the removal of plant material due to accumulations of litter which might cause interference with marsh operation is uncertain as none was apparent during the 4 year study. Although preliminary work in the area of resources recovery for energy or livestock feed is promising, the practice of harvesting to offset expenses would require further process development and economic evaluation.

6.2.3 Pre-treatment of Raw Sewage

Although minimal pre-treatment of the wastewater is desirable to reduce capital and operating costs, the level of pre-treatment was shown to affect effluent quality. The two types of pre-treatment tested at Listowel were found to be inadequate for several reasons.

The complete mix aeration cell is energy intensive and, in this case, did not have enough capacity to sufficiently elevate wastewater oxygen levels. High levels of solids were kept in suspension in the aeration cell and were carried over into the marsh systems. The subsequent sedimentation resulted in a buildup of sludge mats at the influent end of the marshes which interfered with water flow and caused some kill-back of the cattails. On the other hand, lagoon pre-treatment, in these circumstances, represented an inefficient use of land. The lagoon effluent was high in hydrogen sulfide and low in dissolved oxygen during the winter period of ice cover and contained undesirable concentrations of algae in summer.

As a result of the Listowel study, it is possible to recommend pre-treatment options that prevent the problems mentioned above and maintain costs at a reasonable level. A partial-mix facultative aeration cell and phosphorus reduction to about 1.0 mg/L TP would reduce BOD, suspended solids and phosphorus levels entering the marsh. An aeration cell which is 10-12 feet deep with an average detention time of 30 days should provide consistently high oxygen levels to achieve reductions in the concentrations of ammonia and hydrogen sulfide in the marsh, as well as sufficient sludge storage capacity for 20 years. These expectations will be tested at the full-scale facility in Port Perry. The same objectives may be achieved more economically by other means, e.g. a shorter hydraulic detention time and sludge removal without process interruption. However, the latter course would entail sludge haulage costs and the problems of finding suitable sludge disposal sites.

Pre-treatment which includes phosphorus reduction by chemical addition is recommended because of the potential for phosphorus saturation of the marsh sediment. Although conditions in the Listowel marsh systems were not always optimal for phosphorus adsorption, the evidence of a decrease in the soluble P adsorption capacity with cumulative soluble P loadings suggests the advisability of chemically reducing P loadings to the marsh.

6.2.4 Marsh Configuration

The Listowel study demonstrated the influence of marsh design on wastewater treatment. The channelized configuration (channel length width ratio 17:1) showed greater treatment efficiencies than the open marsh design (length width ratio 4:1). System 1, the three-celled system consisting of channelized marsh plus deep pond and open marsh, performed better than the open marsh (System 2) but not as well as the channelized marsh alone (System 3). The channel configuration provided a better distribution of the flow over the marsh surface and prevented short-circuiting which was observed in the open marshes.

The channelized System 4 consistently received higher BOD, SS and TP loadings (on average 3-4 times) than System 2, an open marsh. However, the annual average effluent concentrations of these parameters were comparable with and in some cases lower than those in the effluent of System 2. The installation of baffles in System 2 reduced the short-circuiting which was confirmed with dye studies.

The data indicate the importance of internal flow distribution preferably by earthen berms designed with a high length width ratio (at least 10:1) or, alternatively, other types of barriers to prevent short-circuiting.

6.2.5 Hydraulic Loading and Retention Time

Wastewater renovation in the marsh systems was influenced by the hydraulic retention time. Wastewater must be retained long enough for contaminant removal but an excessively long retention results in stagnation and anoxia. At a constant hydraulic loading, environmental factors such as evapotranspiration in summer and ice formation in winter affected the wastewater residence time. Evapotranspiration increased retention by reducing flows whereas ice formation decreased retention due to the reduction in the operational marsh volume.

Marsh design permitted the control of retention time through the adjustment of hydraulic loading and water depth. Although flow rates were changed during the study, the only lever for the adjustment of retention time under full-scale operational conditions is the regulation of water depth. Raising the water depth to $\geq$30 cm prior to winter and reducing the water depth to $\leq$10 cm in summer was generally effective in regulating retention times. The outflow structures should he designed without a lip to allow complete draining of the marsh if required in the event of summer stagnation. The latter conditions increase the retention time beyond the optimum range and are associated with elevated levels of ammonia, hydrogen sulfide and soluble phosphorus.

The Listowel data suggest that 7-14 days are required for effective wastewater treatment. Treatment efficiencies were reduced when the retention time deviated from this range. Short retention times (<7 days) resulted from short-circuiting in the open marshes and ice formation particularly noticeable during the first year in all systems when the flow rate was increased. Retention times greater than 14 days occurred primarily in the summer months when evapotranspiration rates were high and anoxic conditions prevailed. This was particularly pronounced during the summer of 1983.

Pre-treated wastewater was applied to the channelized systems at hydraulic loading rates in the range of 130-200 m^3/ha/d for most of the 4 year period during which substantial reductions in wastewater contaminants were achieved. During the first year when hydraulic loadings were increased to 260-530m^3/ha/d, treatment efficiencies declined. The annual BOD loadings to the marsh systems depended on the type of sewage pre-treatment. Lagoon-treated influent to the marshes ranged between 75 and 100 g/m^2, whereas the BOD_5 loading range in the aeration cell effluent was 250 to 350 g/m^2.

6.2.6 Nuisance Organisms

The study of nuisance insects breeding in the marsh focused on the major species of concern, midges and mosquitoes. The number of midges were found to be small compared to the numbers at the site of conventional lagoons. Although large numbers of mosquitoes were produced at the marsh, the species collected do not feed on humans. Several species are of significance as vectors of encephalitis. It is therefore recommended that in the event of encephalitis activity, efforts be made to reduce mosquito numbers. The application of the bacterial pesticide Bacillus thuringiensis var. isrealiensis was shown to be an effective mosquito control technique. Controlling vegetation on the berms was suggested as a method of reducing open water areas which function as egg laying sites.

6.3 Implementation Options

Artificial marsh treatment of domestic wastewater in rural areas and small communities is only economically competitive with conventional seasonal retention lagoons if the marsh system can operate with year-round continuous discharge. Storage facilities would increase costs considerably. An important consideration in choosing marshland treatment is the treatment goals for a potential site. These should be based on Ministry objectives for surface water

quality, as well as the site specific hydrology, water quality and uses of the receiving waters. Continuous discharge requires a consistently high effluent quality. This is particularly critical during periods of reduced stream flow and low dilution as often occurs in Ontario streams in summer. The installation of marshland wastewater treatment facilities may therefore be restricted by the sensitivity of the receiving water body.

Based on the results of the analysis of Listowel effluent data, marsh treatment is capable of producing effluent of secondary to tertiary quality with respect to BOD, SS and TP. Calculated from the highest monthly ammonia level in the marsh effluent in summer, flows in the receiving waters should provide a dilution of 10:1 to meet the Ministry surface water objective of .02 mg/L for un-ionized ammonia. Although the monthly average un-ionized ammonia levels in the marsh effluent of System 3 were low ($\leq$ .045 mg/L), the objective is compared with the summer maximum monthly average un-ionized ammonia concentration which would result in the possible event of an elevated pH (i.e. pH 8) in the receiving waters.

However, conditions at Listowel were experimental and did not provide the design and operational conditions for achieving optimum contaminant removal for this type of sewage treatment system. The recommended pre-treatment process and other marsh design and management recommendations (described in Section 6.2) are based on experiences gained at Listowel. These have been incorporated into a full-scale marshland sewage treatment demonstration facility at Port Perry and it is expected that these modifications will provide an improved treatment capability. The advantage of the pre-treatment unit constructed at Port Perry is a marsh effluent with a combination of consistently high oxygen levels, low solids, hydrogen sulfide and phenols concentrations and lower bacterial levels (due to longer retention time) than produced by pre-treatment methods tested at Listowel. The effluent quality produced at Port Perry will provide a better indication of the potential for the application of this type of marshland sewage treatment process in Ontario.

Marshland treatment of municipal wastewater may be an option considered in the following circumstances.

a) Provision of New Wastewater Treatment Facilities: Many unserviced rural communities experience difficulties with individual septic tank systems. Problems typically arise because of inadequately sized lots, improper soils or following installation of communal waterworks which generally increase water usage. Although correction of problems on an individual basis is preferrable, development of a new communal facility may, in some circumstances, be unavoidable. Similar applications for marshland technology include campgrounds, resorts, large agricultural operations or industries (i.e. food processing) which require "on site" wastewater treatment facilities.

b) Conversion of Seasonal Retention Lagoons: Since marshland systems require less land than seasonal retention lagoons, the conversion of existing lagoons to marshes would provide increased hydraulic capacity without further land aquisition. However, very low summer dilution capacity might prohibit release of effluent during summer months.

c) Conversion of Continuous Discharge Lagoons: Since land requirements for continuous discharge lagoons and marsh systems are comparable, conversion from lagoon to marsh will not provide increased hydraulic capacity. The major benefit would be an improved effluent quality to meet increasingly more stringent receiving water criteria.

d) Effluent Polishing for Primary and Secondary Treatment Plants: The addition of a marsh for final effluent polishing could upgrade primary effluent to secondary or improve secondary effluent. This option, however, is limited by the availability of land or a natural wetland in close proximity to the sewage treatment plant.

6.4 Need for Further Study

The results of the Listowel pilot study were translated into design and operational recommendations which require testing in a full-scale marsh treatment facility, as has been established by the Ministry at Port Perry, Ontario. There are always some uncertainties associated with the transfer of pilot study results to full-scale application. The Port Perry demonstration facility will be used to assess the treatment capabilities and operating procedures under fluctuating hydraulic loadings and variations in sewage strength not encountered at Listowel. These result from the impact of storm events on the volume and content of domestic wastewater generated by a community.

In addition, the operation of the Listowel marsh systems indicated serious shortcomings in the two methods used to pre-treat the raw sewage. Based on these findings, an improved pre-treatment process was recommended. An assessment of the efficiency of the recommended pre-treatment requires the measurement of oxygen concentrations in the aeration cell effluent and the determination of whether these concentrations reduce ammonia, hydrogen sulfide and other parameters to acceptable levels in the marsh effluent. Sludge accumulation in the pre-treatment aeration cell should be monitored to more accurately assess the design life of the aeration cell.

The adequacy of flow distribution in the wide channel design of the full-scale marsh treatment facility and the rate of plant litter accumulation in the marsh should also be determined. The presence of a demonstration facility will contribute to the further fine-tuning of system guidelines and will assist in technology transfer. The labor requirements and costs of running a full-scale system would be useful in assessing the economic competitiveness of marshland sewage treatment.

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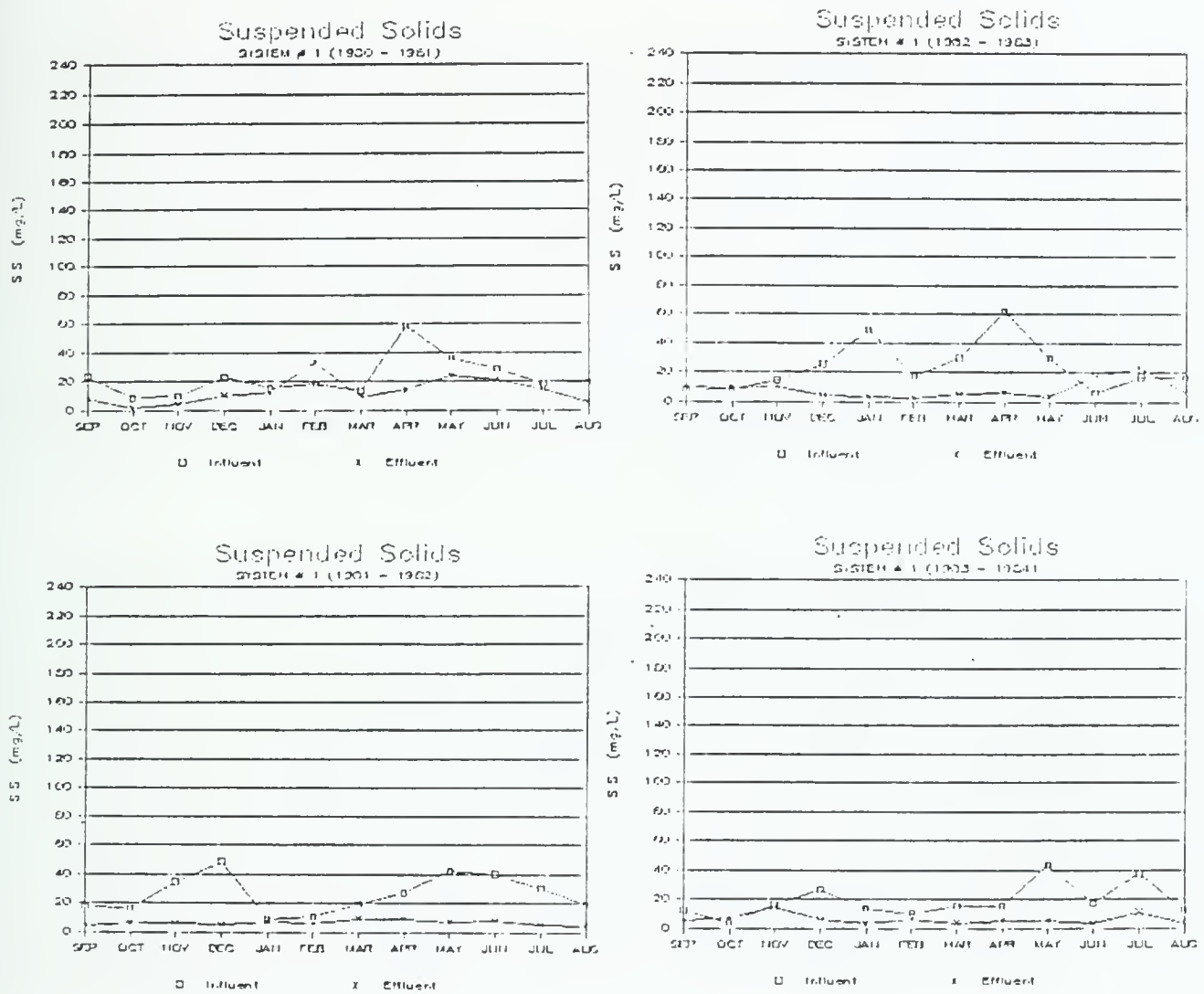
APPENDIX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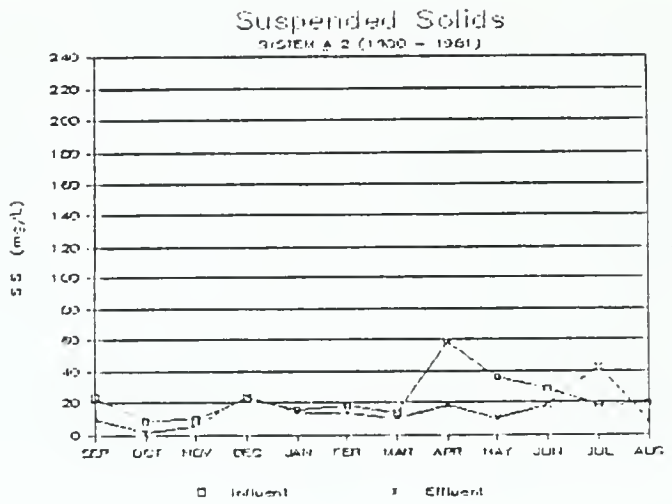
SYSTEM # 1 (1980 - 1981)
Suspended Solids
SYSTEM # 1 (1982 - 1983)
Suspended Solids
SYSTEM # 1 (1981 - 1982)
Suspended Solids
SYSTEM # 1 (1983 - 1984)
SS (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
□ Influent
x Effluent

Figure A.1 Monthly average suspended solids concentrations in Systems 1-5 (4 years).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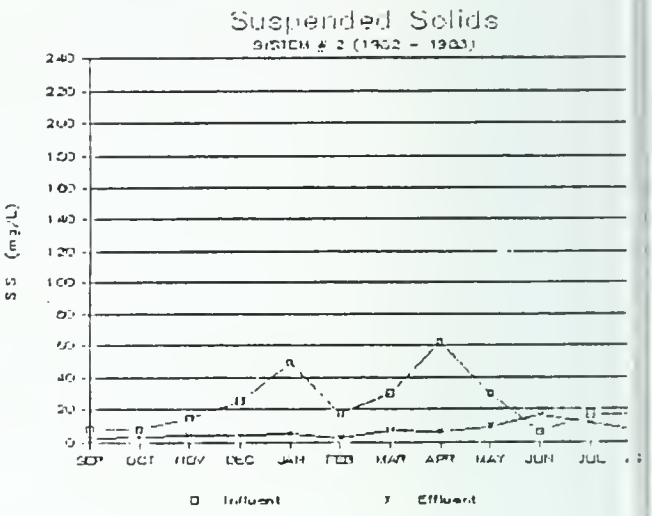
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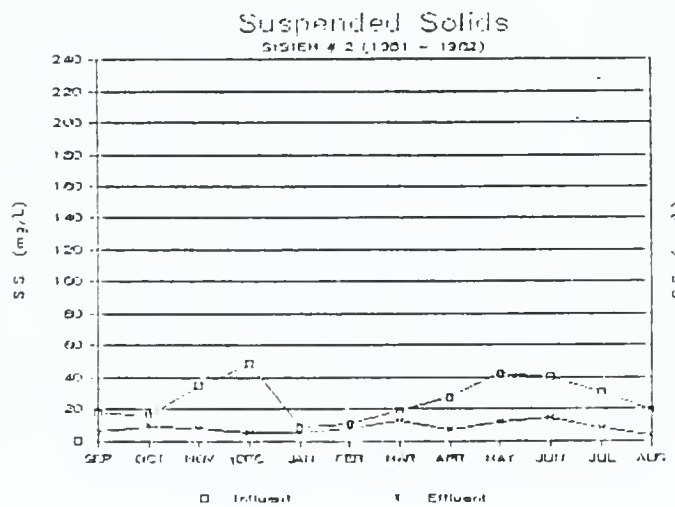
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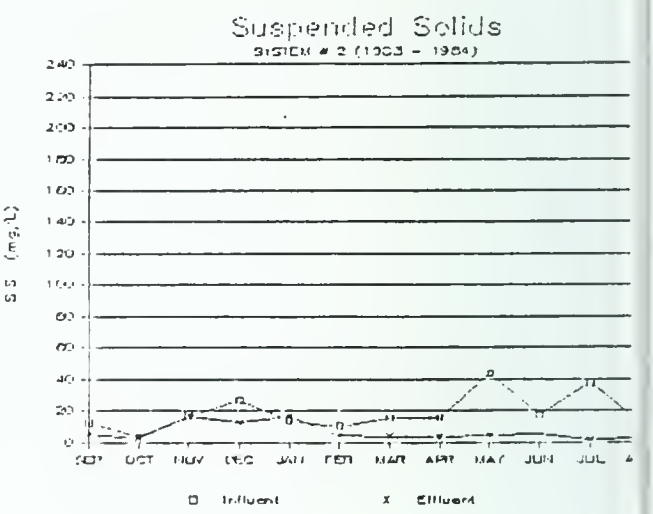
Influent
Effluent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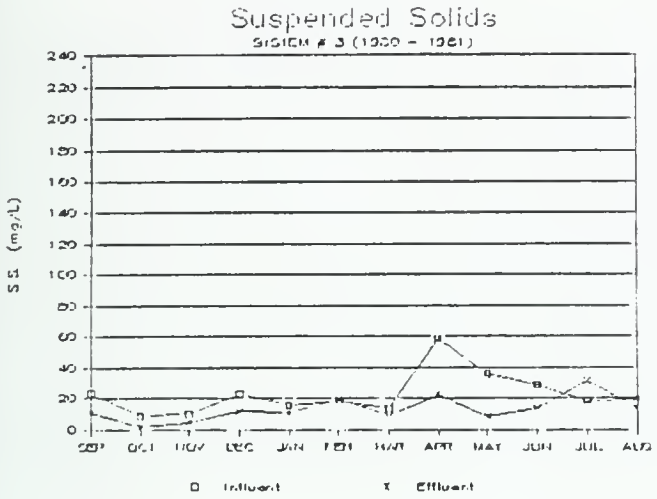
Influent
Effluent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200
180
160
140
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100
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20
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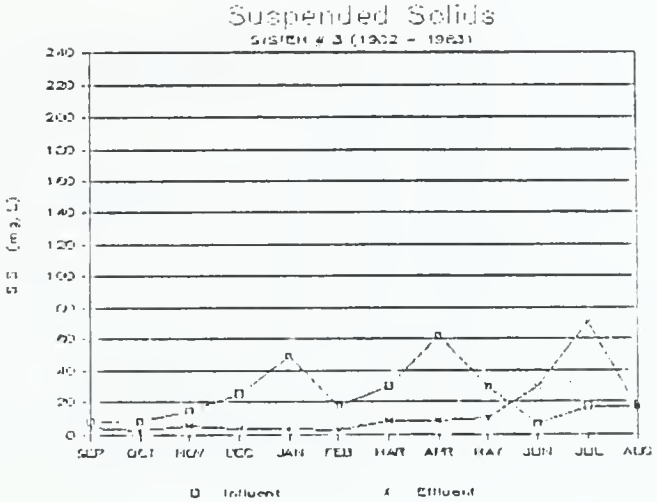
O Influent
X Effluent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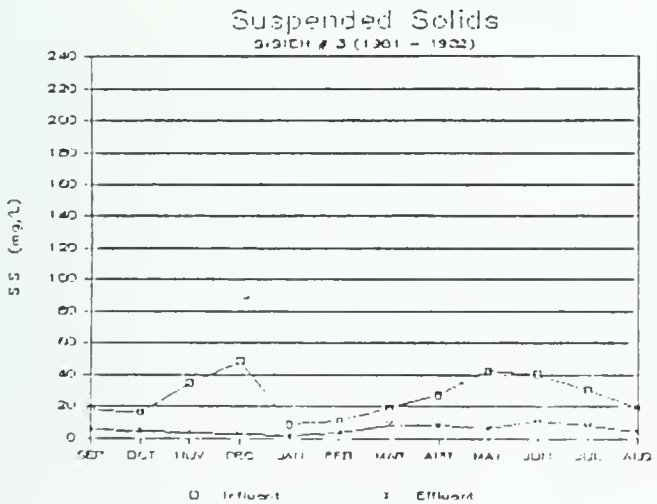
O Influent
X Effluent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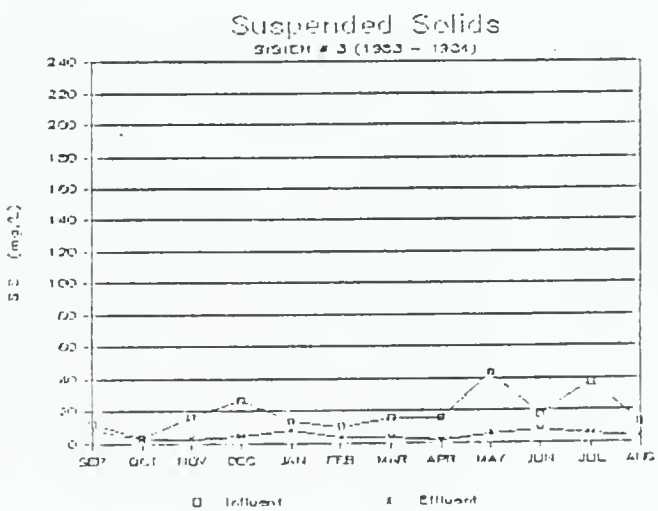
O Influent
X Effluent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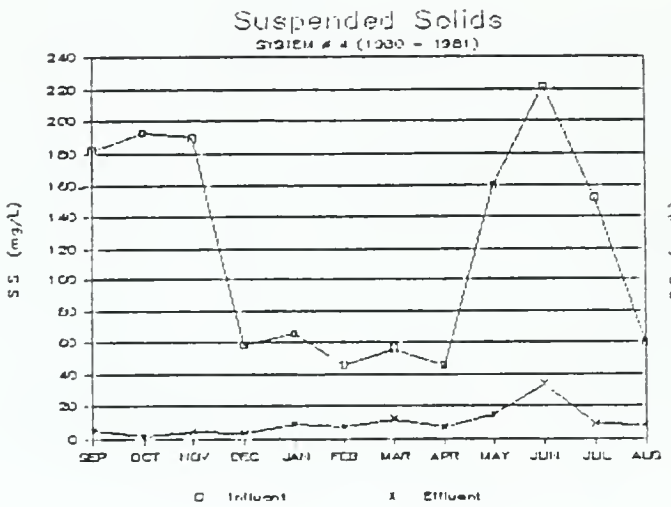
O Influent
X Effluent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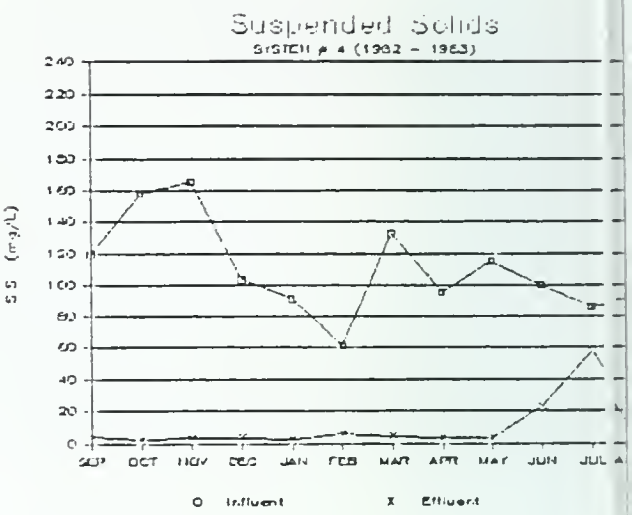
O Influent
X Effluent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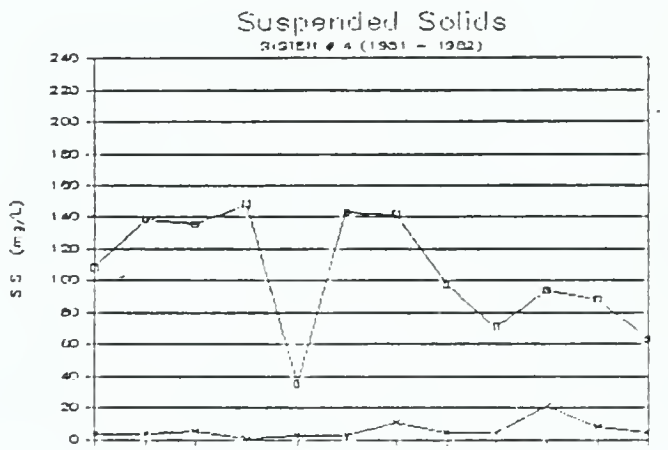
O Influent
X Effluent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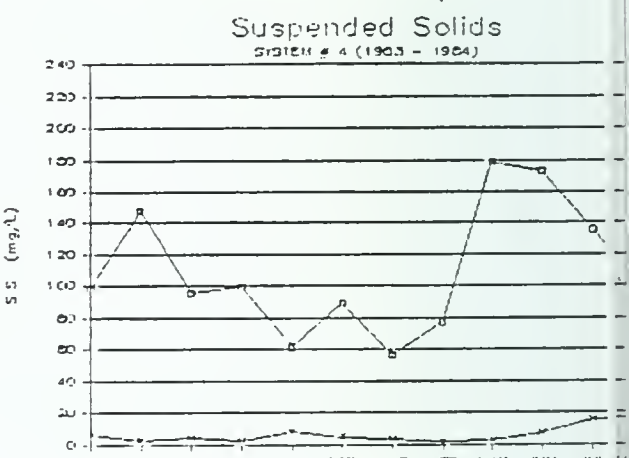
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent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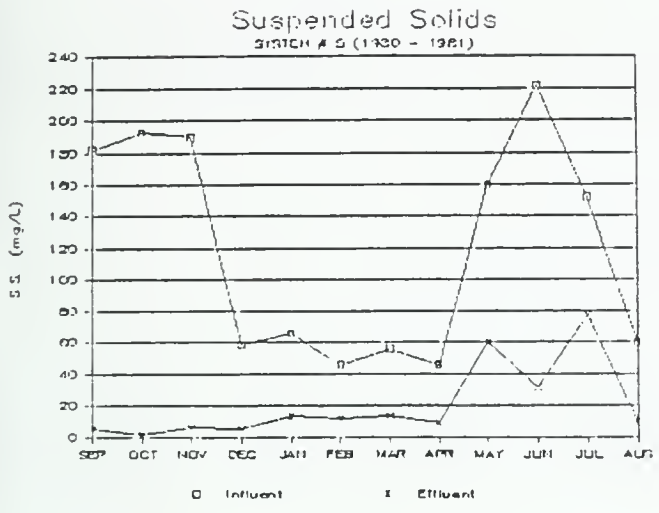
O Influent
X Effluent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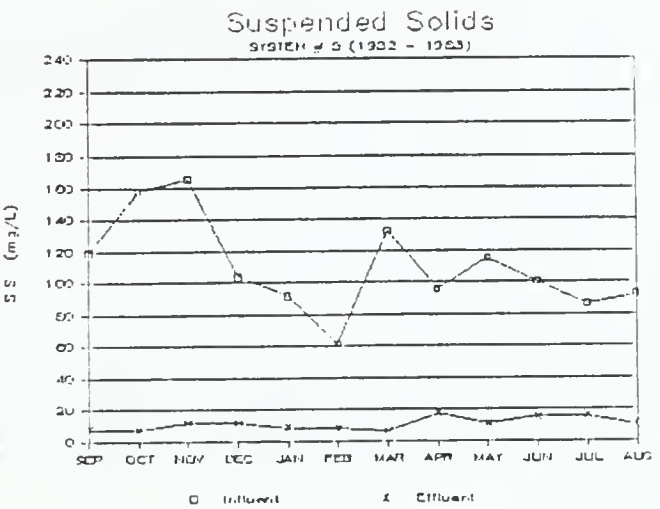
o Influent
x Effluent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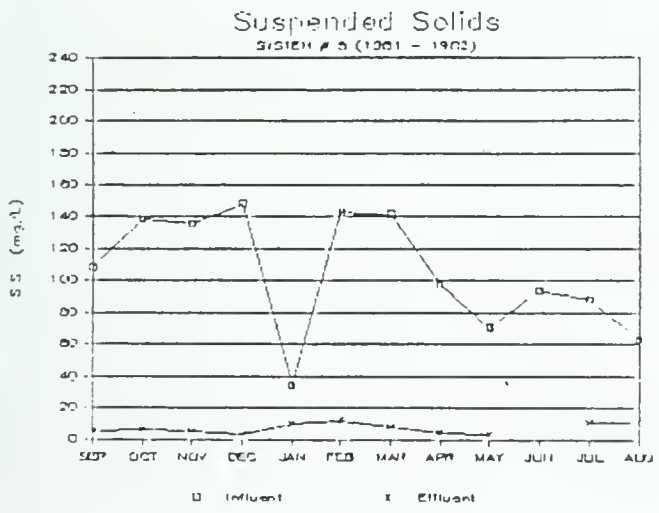
S S (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent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S S (mg/L)
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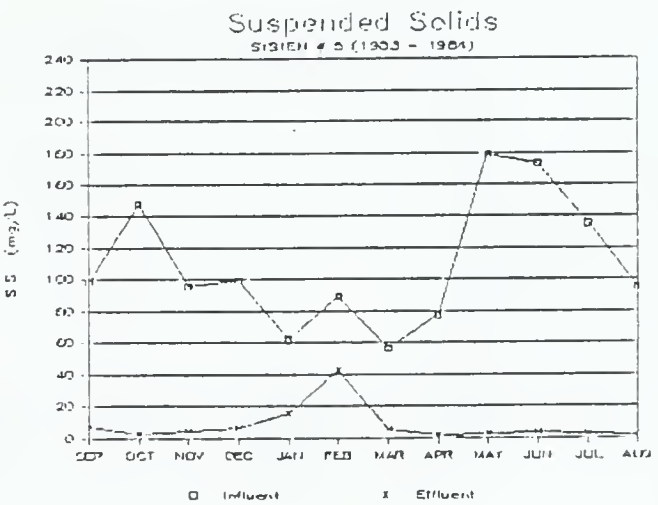
o Influent
x Effluent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o Influent
x Effluent

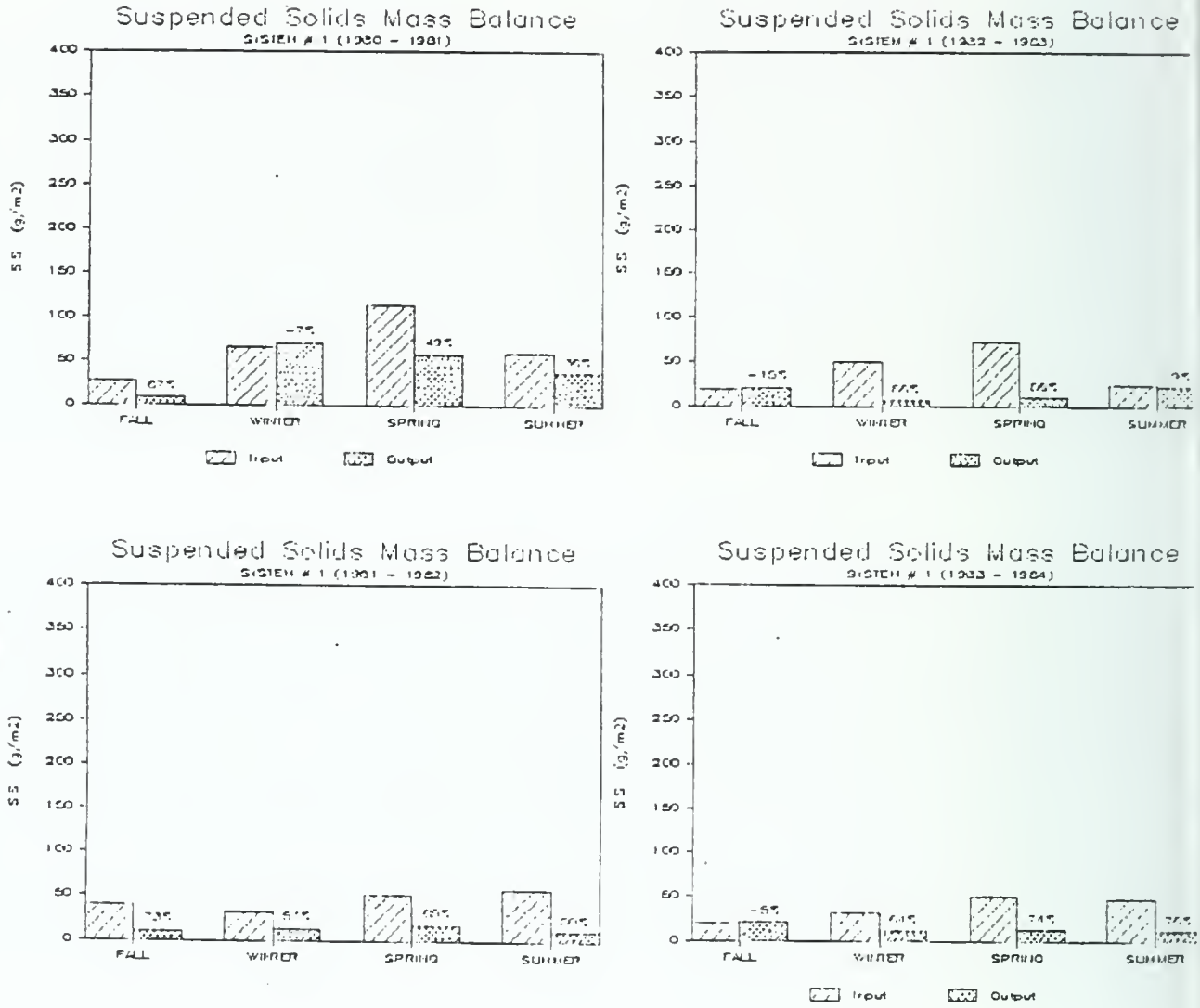
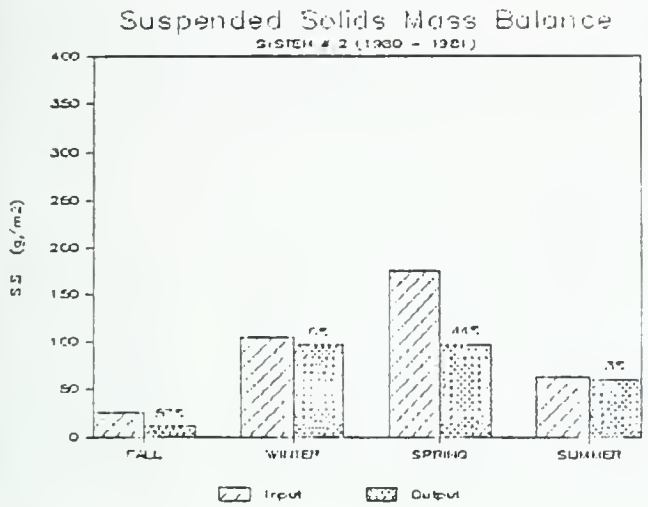


Figure A.2 Seasonal suspended solids mass balances and removal efficiencies in Systems 1-5 (4 years).


Suspended Solids Mass Balance
SYSTEM # 2 (1980 - 1981)
SS (g/m2)
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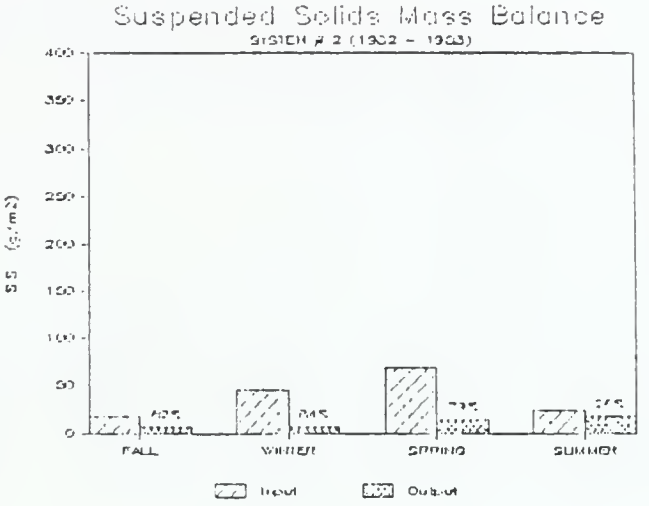
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 2 (1982 - 1983)
SS (g/m2)
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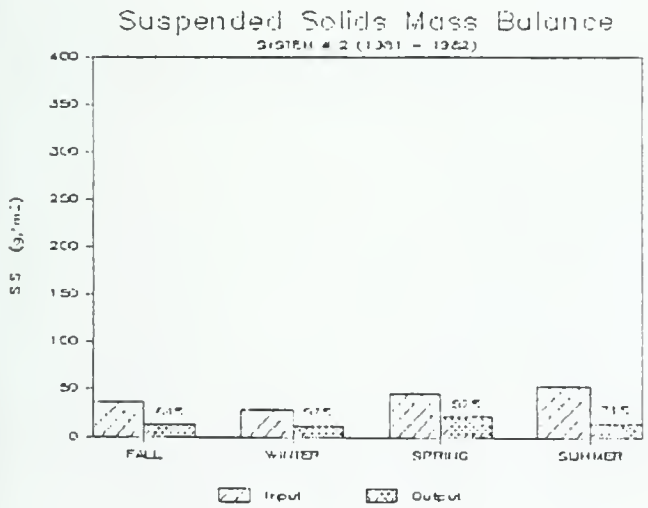
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output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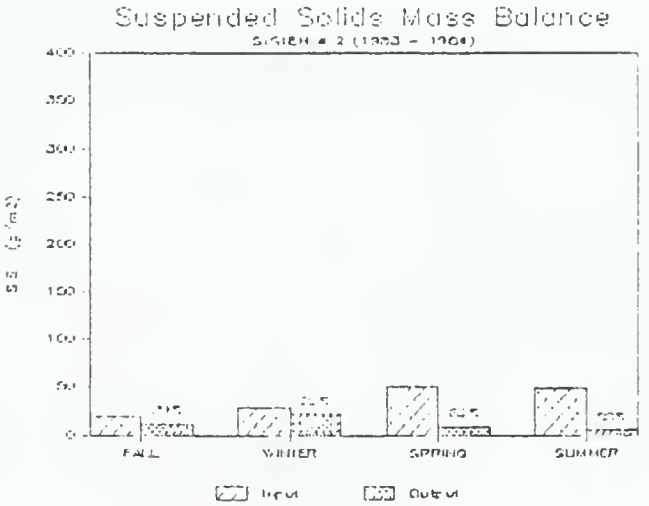
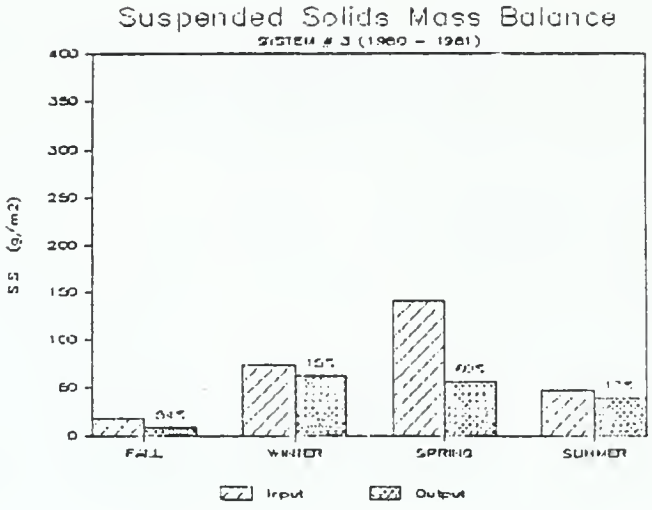
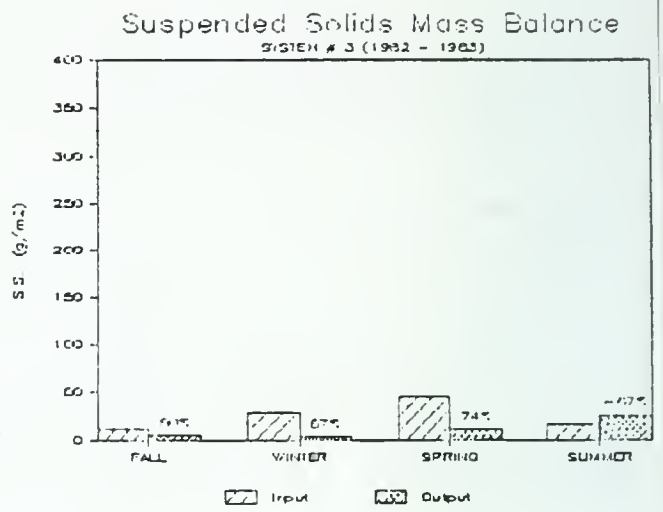
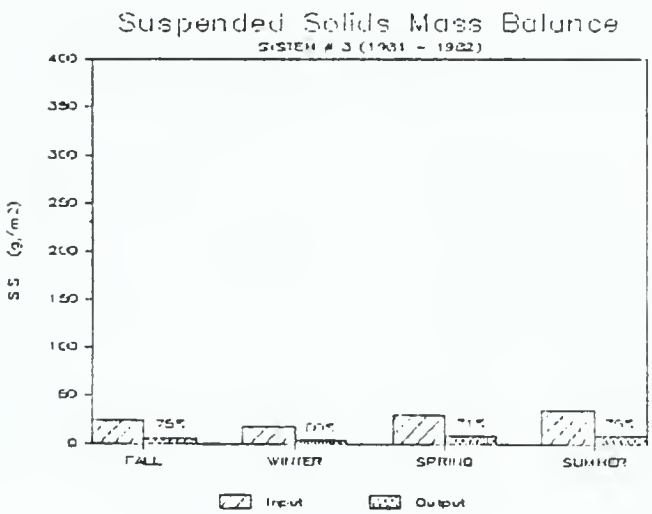
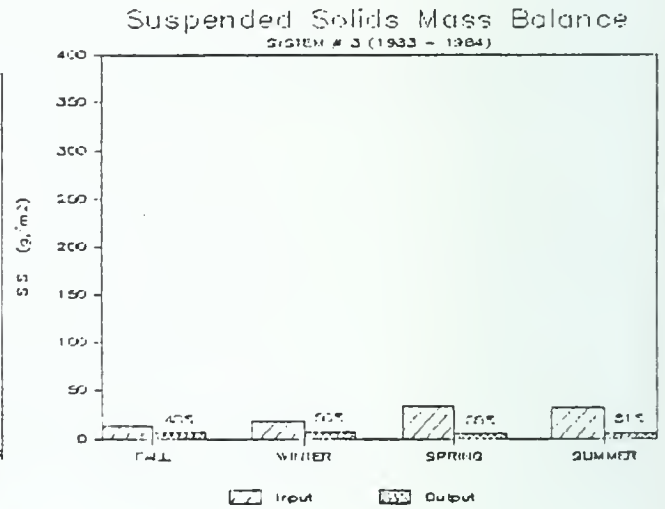
SYSTEM # 2 (1981 - 1982)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

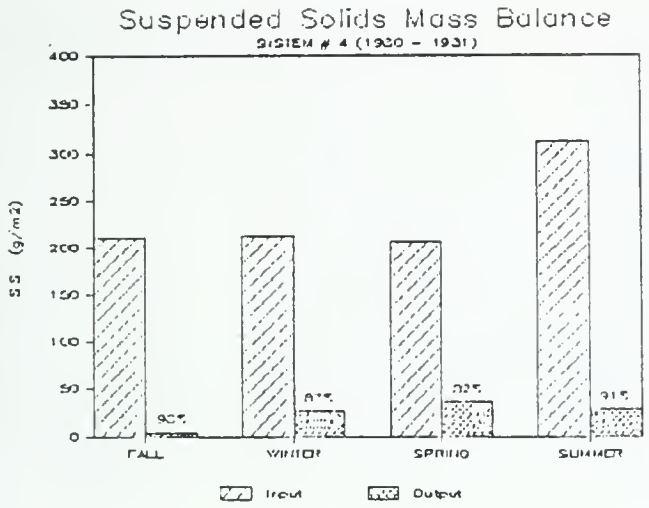

Suspended Solids Mass Balance
SYSTEM # 2 (1983 - 1984)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

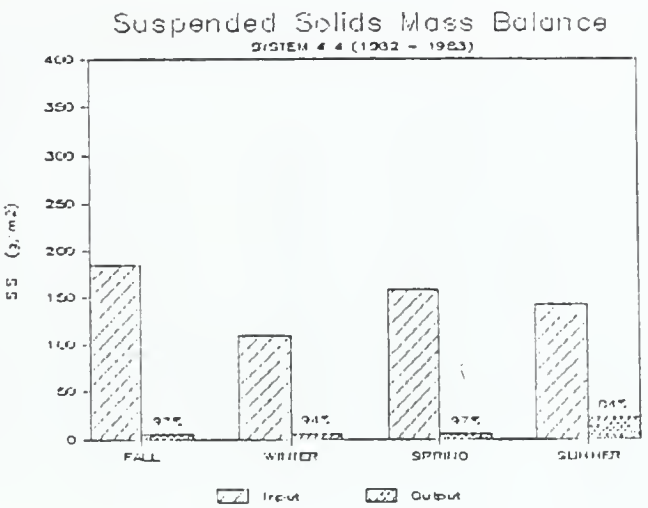

Suspended Solids Mass Balance
SYSTEM # 3 (1980 - 1981)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

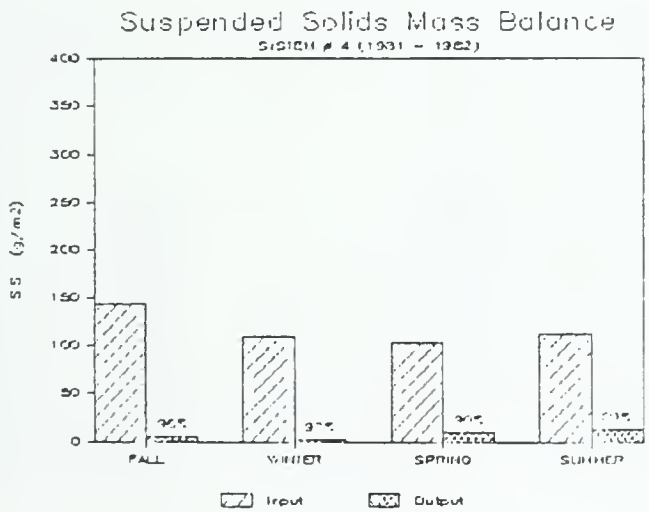

Suspended Solids Mass Balance
SYSTEM # 3 (1982 - 1983)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

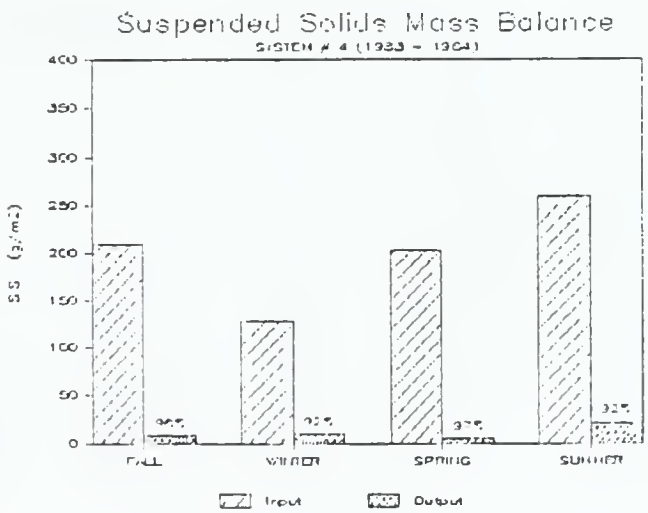

Suspended Solids Mass Balance
SYSTEM # 3 (1981 - 1982)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

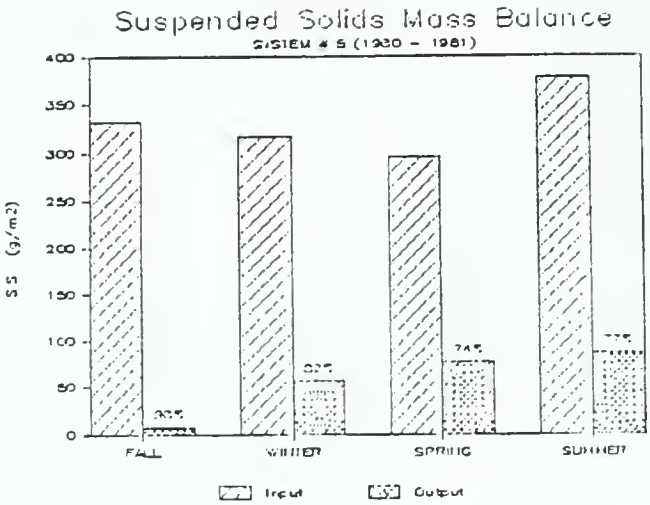

Suspended Solids Mass Balance
SYSTEM # 3 (1983 - 1984)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

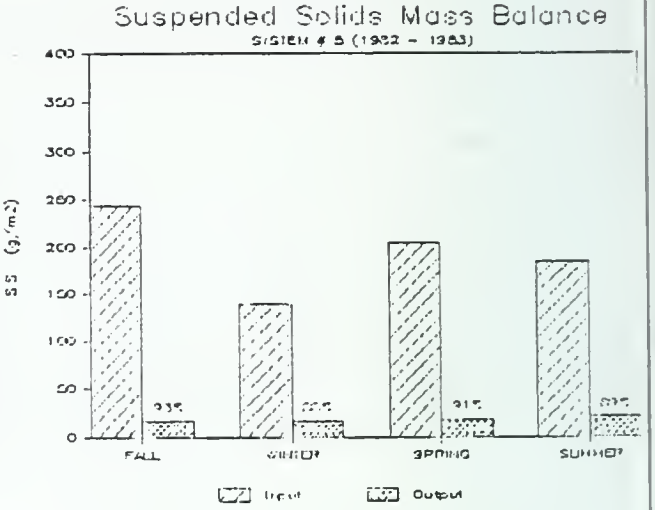

Suspended Solids Mass Balance
SYSTEM # 4 (1980 - 1981)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

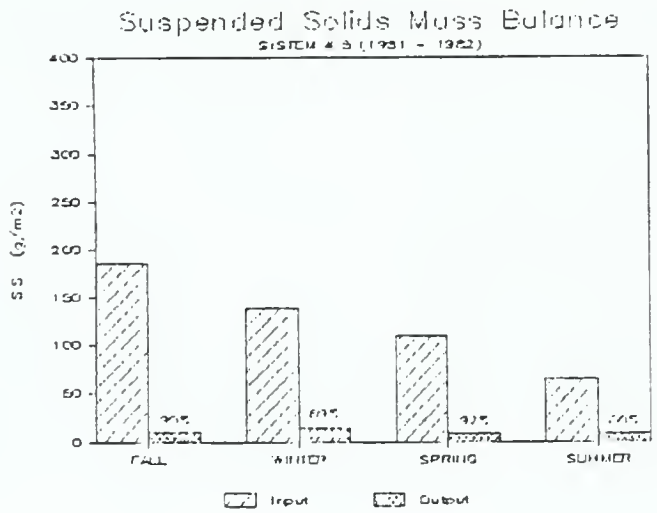

Suspended Solids Mass Balance
SYSTEM # 4 (1982 - 1983)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

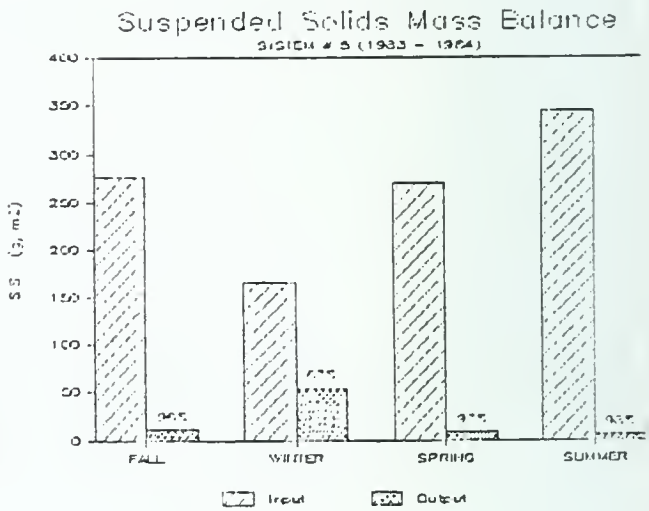

Suspended Solids Mass Balance
SYSTEM # 4 (1981 - 1982)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 4 (1983 - 1984)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 5 (1980 - 1981)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 5 (1982 - 1983)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 5 (1981 - 1982)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output


Suspended Solids Mass Balance
SYSTEM # 5 (1983 - 1984)
SS (g/m2)
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

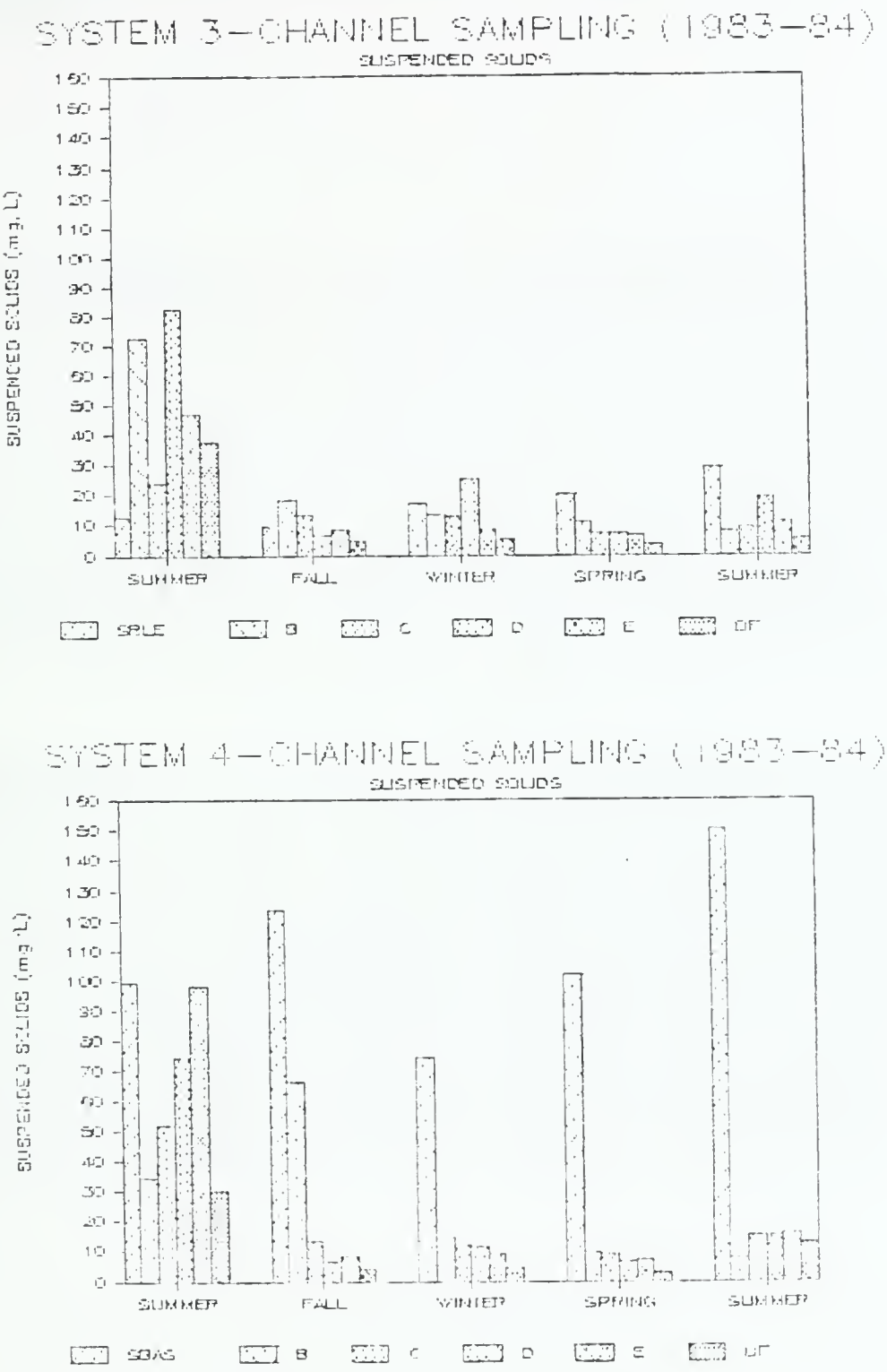


Figure A.3 Seasonal suspended solids concentrations in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of Systems 3 and 4 (1983-4).

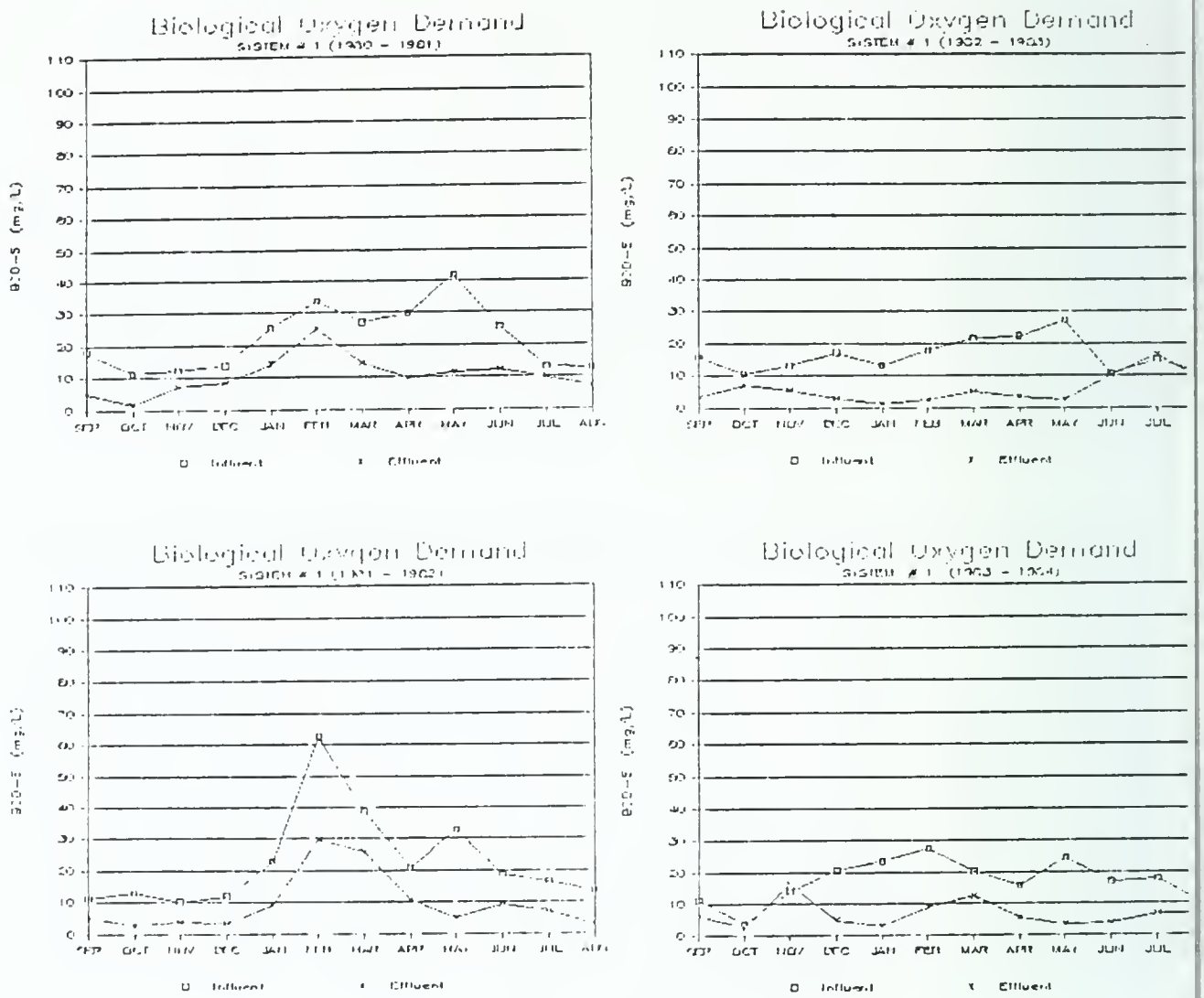
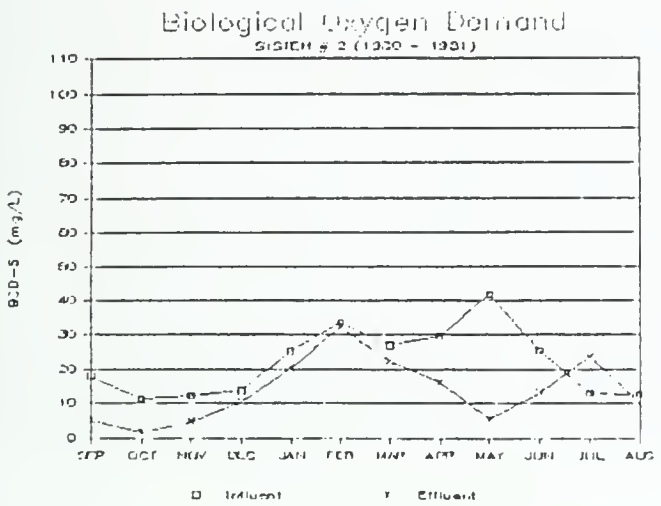
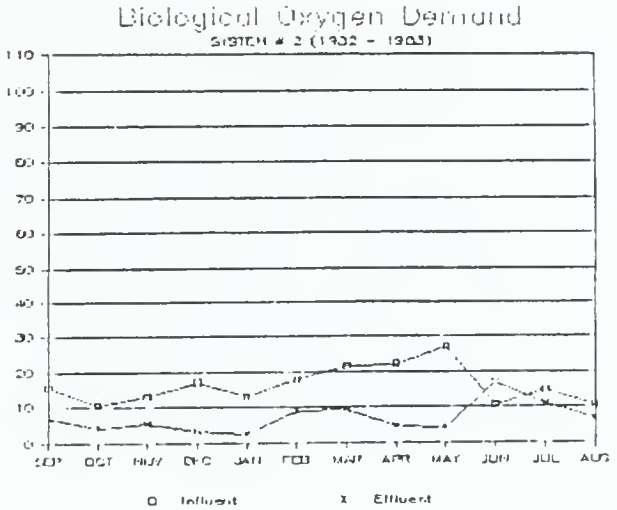
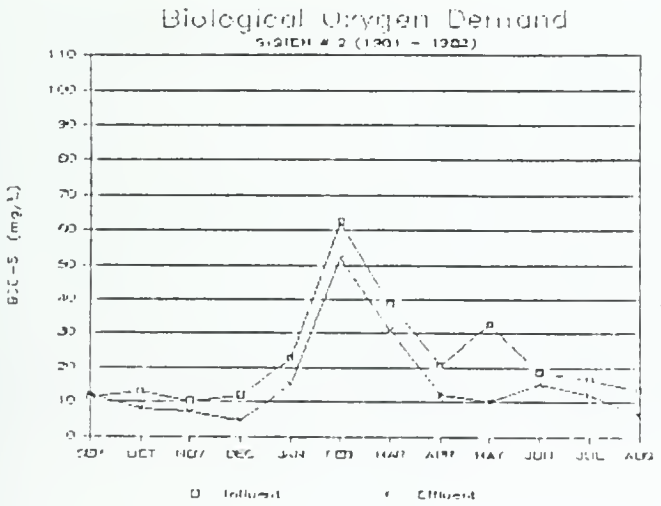
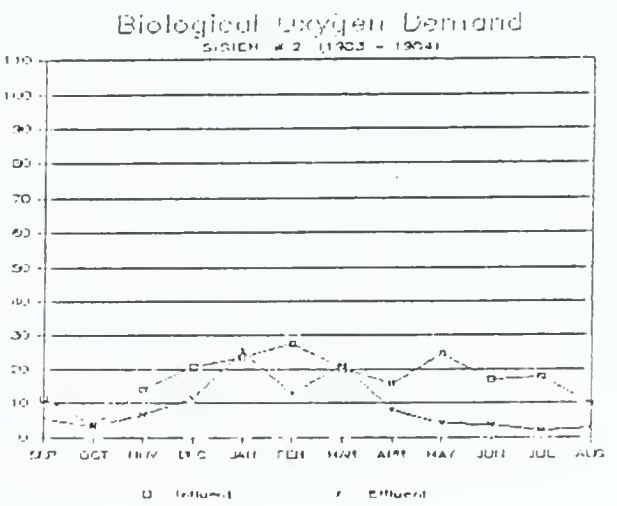

Biological Oxygen Demand
SYSTEM # 1 (1980 - 1981)
Biological Oxygen Demand
SYSTEM # 1 (1982 - 1983)
Biological Oxygen Demand
Biological Oxygen Demand
B.O.D. (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

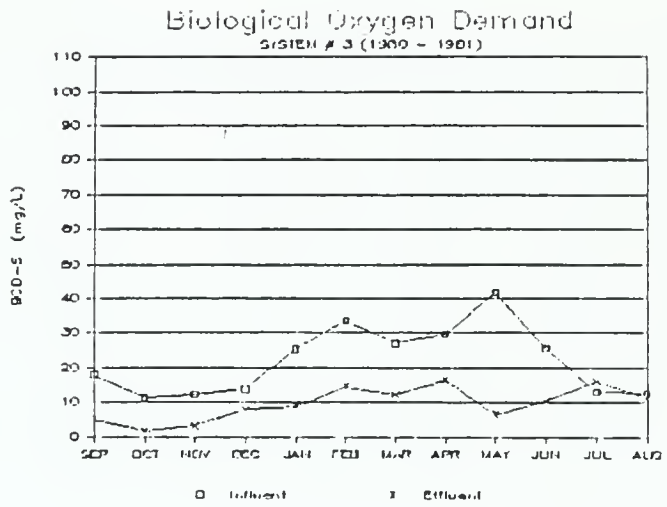
Figure A.4 Monthly average BOD_5 levels in Systems 1-5 (4 years).

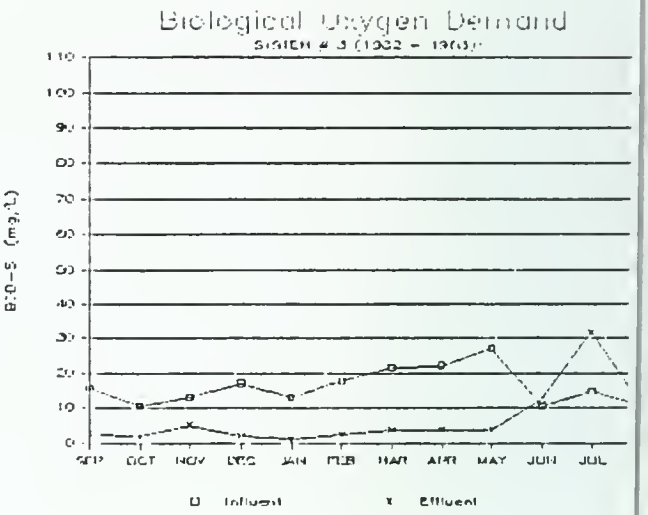

Biological Oxygen Demand
SIGIEN # 2 (1930 - 1931)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

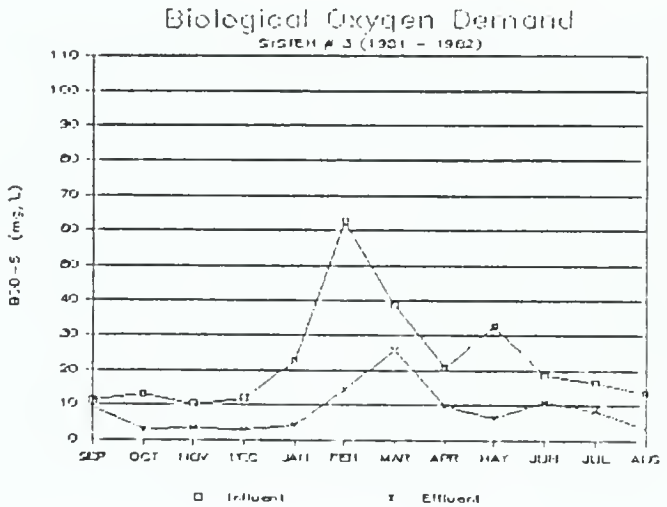

Biological Oxygen Demand
SIGIEN # 2 (1932 - 1933)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

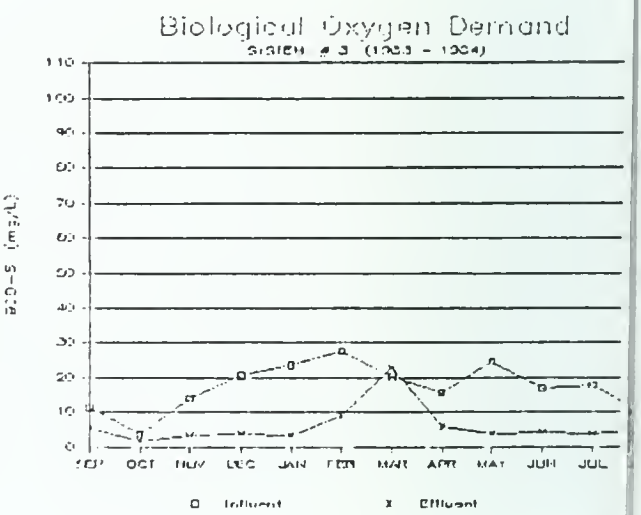

Biological Oxygen Demand
SIGIEN # 2 (1931 - 1932)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

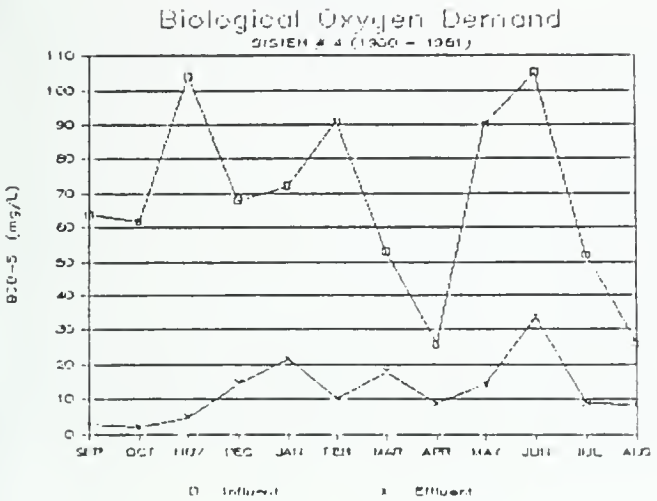

Biological Oxygen Demand
SIGIEN # 2 (1933 - 1934)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

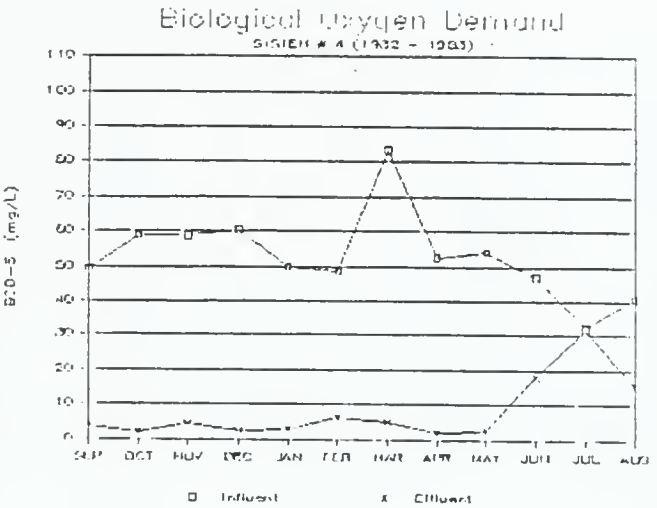

Biological Oxygen Demand
SISIEN # 3 (1980 - 1981)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

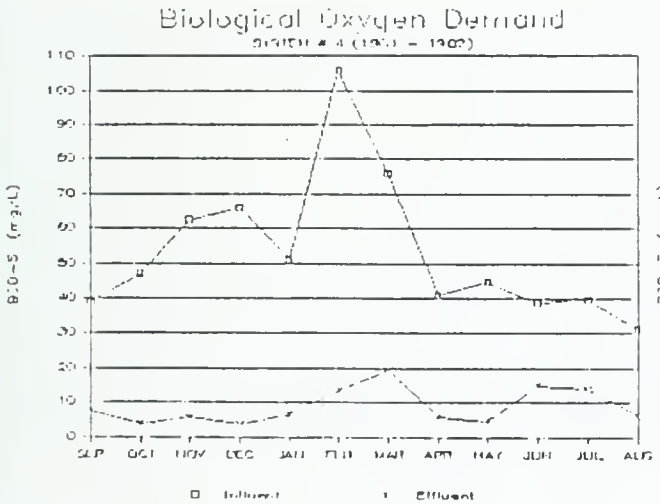

Biological Oxygen Demand
SISIEN # 3 (1982 - 1983)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

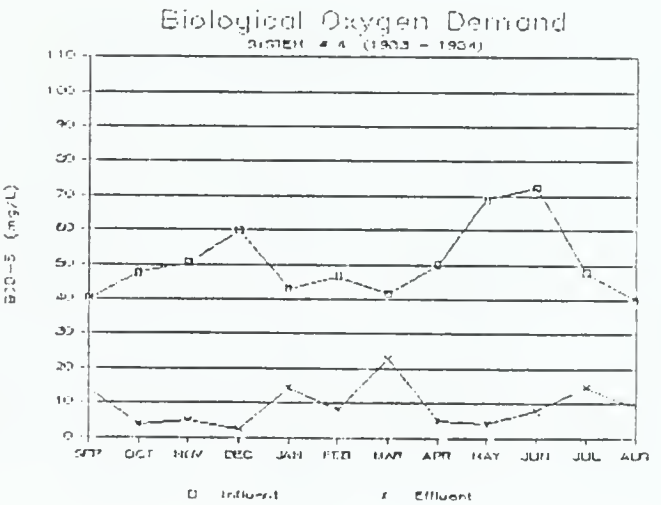

Biological Oxygen Demand
SISIEN # 3 (1981 - 1982)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

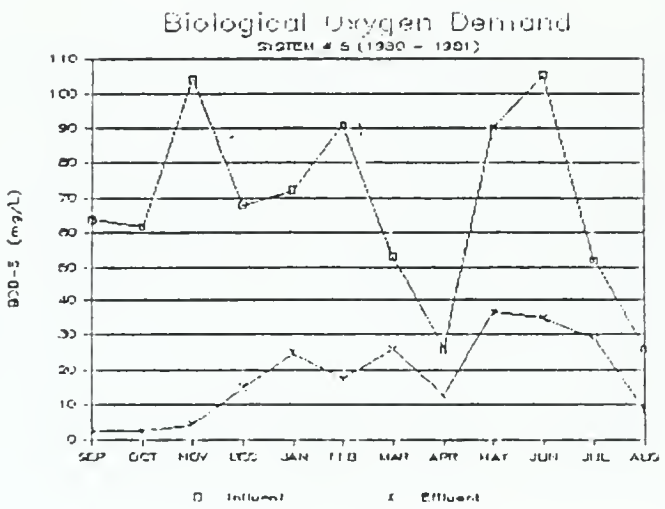

Biological Oxygen Demand
SISIEN # 3 (1983 - 1984)
B.O.D-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

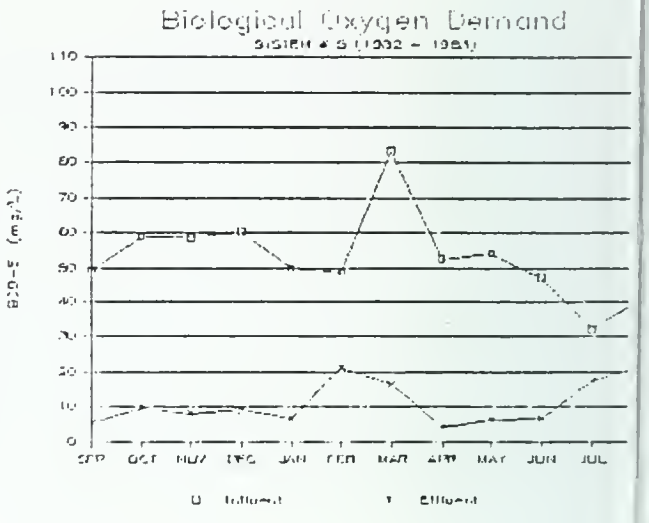

Biological Oxygen Demand
SISIEN # 4 (1960 - 1961)
B:O-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

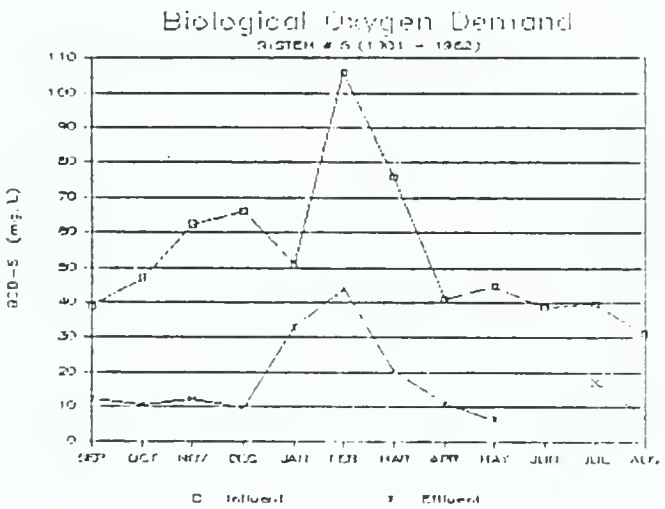

Biological Oxygen Demand
SISIEN # 4 (1932 - 1933)
B:O-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

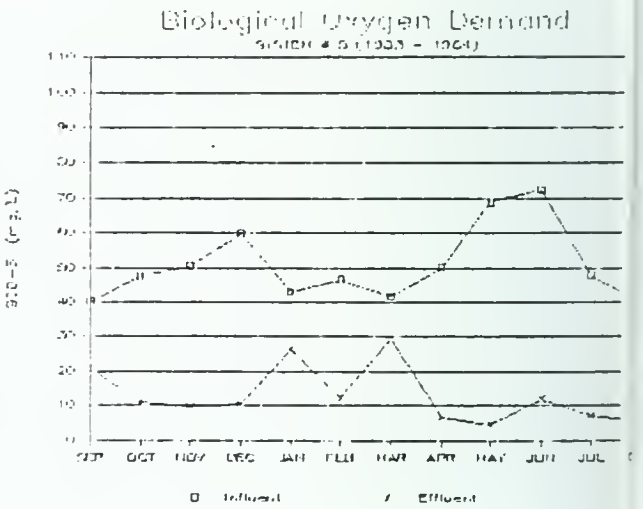

Biological Oxygen Demand
B:O-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Biological Oxygen Demand
SISIEN # 4 (1933 - 1934)
B:O-5 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Biological Oxygen Demand
SYSTEM # 5 (1980 - 1981)
BOD-5 (mg/L)
110
100
90
80
70
60
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Biological Oxygen Demand
SYSTEM # 5 (1982 - 1983)
BOD-5 (mg/L)
110
100
90
80
70
60
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent


Biological Oxygen Demand
SYSTEM # 5 (1981 - 1982)
BOD-5 (mg/L)
110
100
90
80
70
60
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Biological Oxygen Demand
SYSTEM # 5 (1983 - 1984)
BOD-5 (mg/L)
110
100
90
80
70
60
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

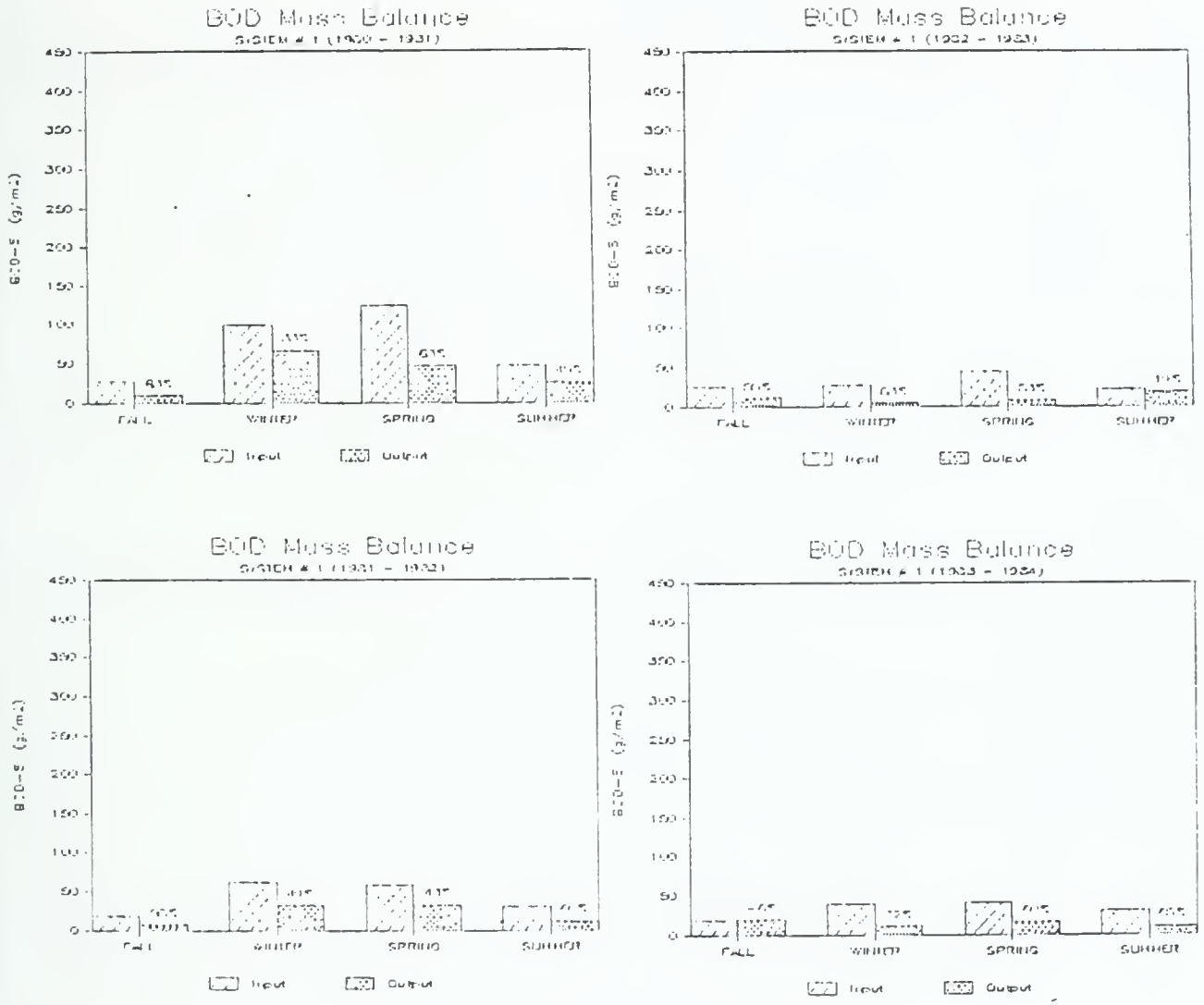
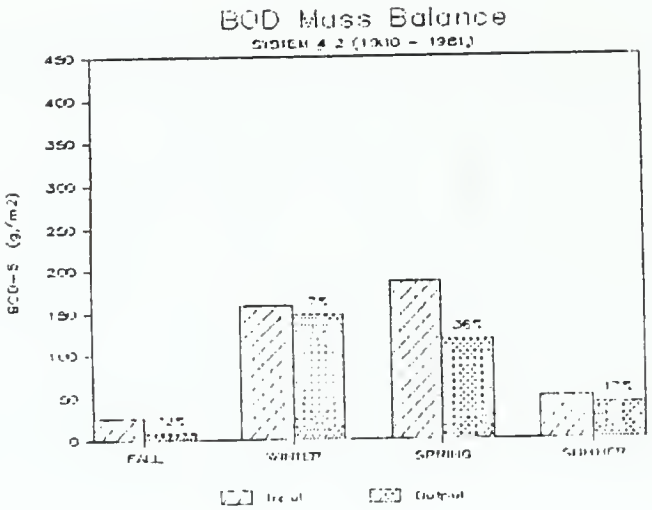
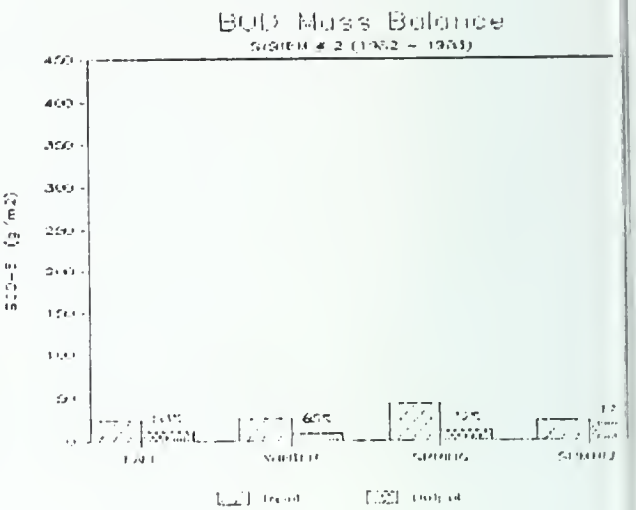
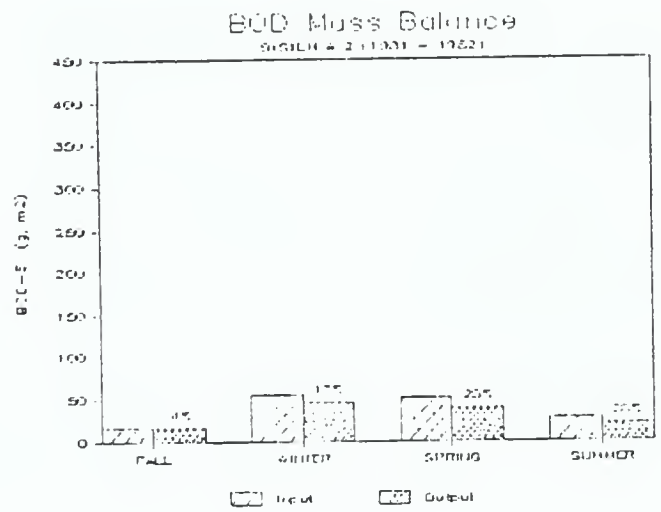
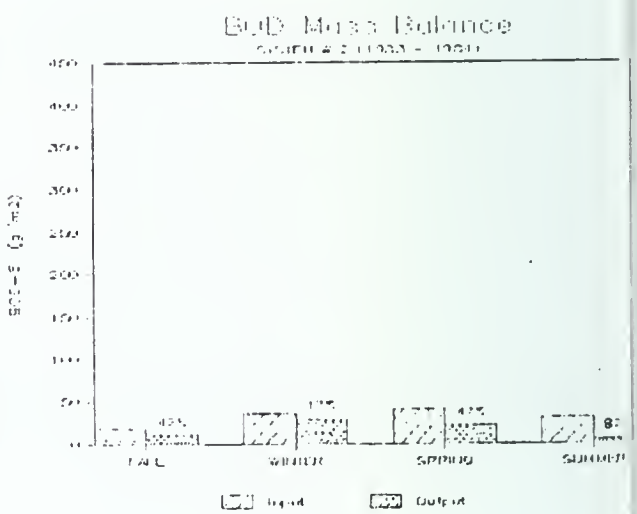


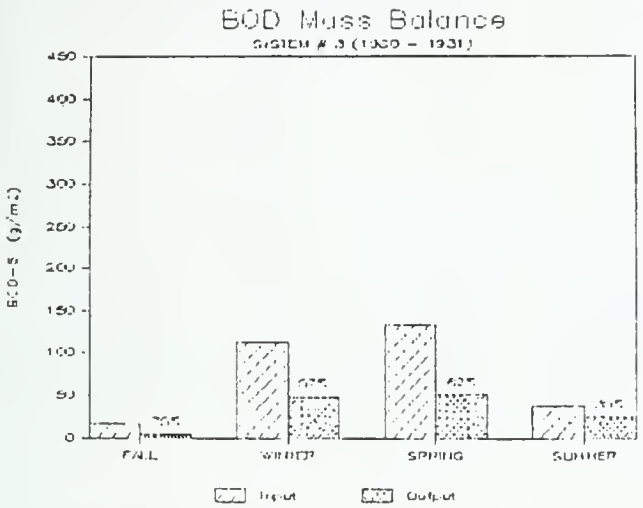
Figure A.5 Seasonal BOD_5 mass balances and removal efficiencies in Systems 1-5 (4 years).

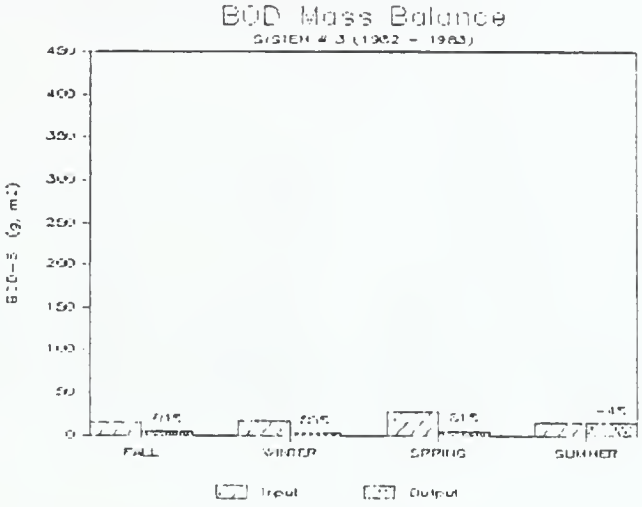

BOD Mass Balance
SYSTEM # 2 (1980 - 1981)
BOD-5 (g/m2)
450
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

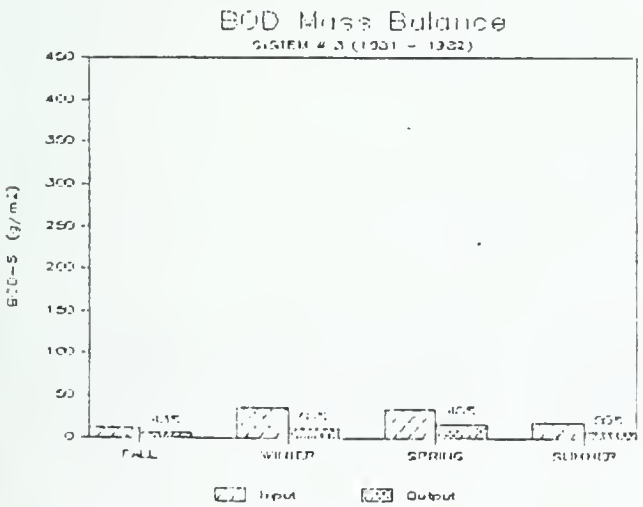

BOD Mass Balance
SYSTEM # 2 (1982 - 1983)
BOD-5 (g/m2)
450
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

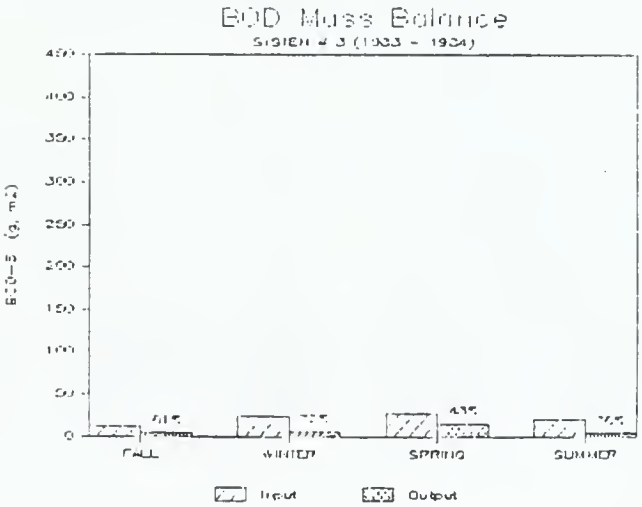

BOD Mass Balance
SYSTEM # 2 (1981 - 1982)
BOD-5 (g/m2)
450
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

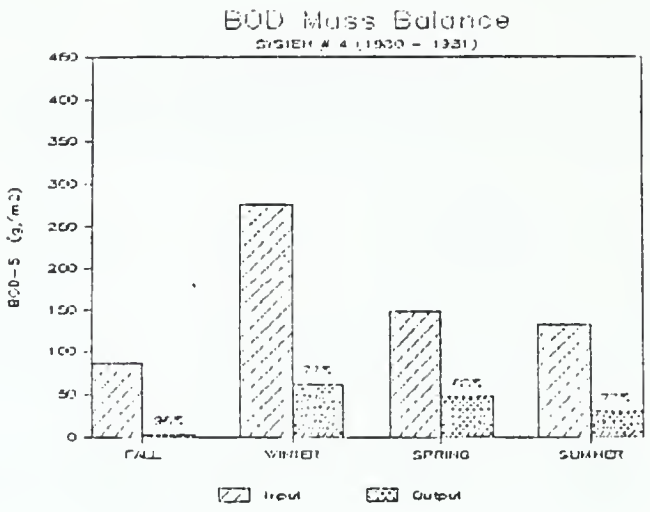

BOD Mass Balance
SYSTEM # 2 (1983 - 1984)
BOD-5 (g/m2)
450
400
350
300
250
200
150
100
50
0
FALL
WINTER
SPRING
SUMMER
Input
Output

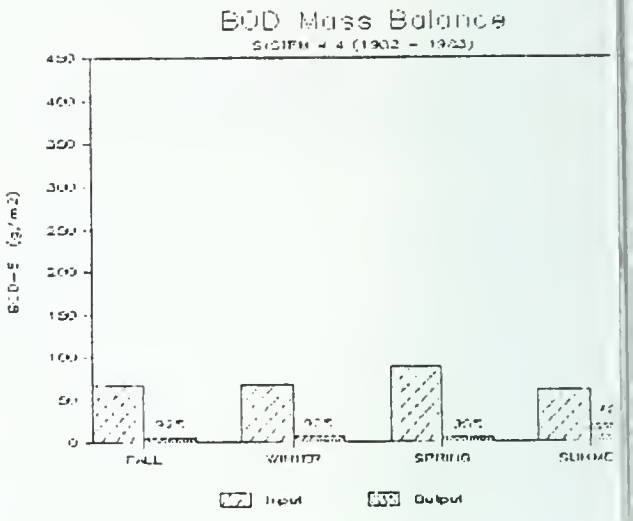

BOD Mass Balance
SYSTEM # 3 (1930 - 1931)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

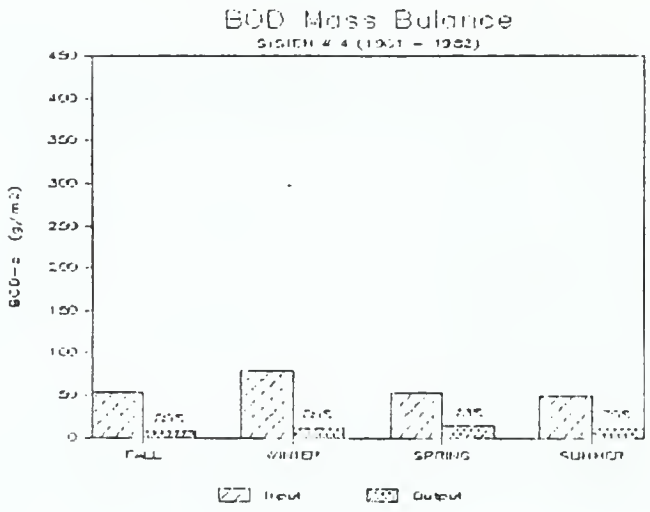

BOD Mass Balance
SYSTEM # 3 (1932 - 1933)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

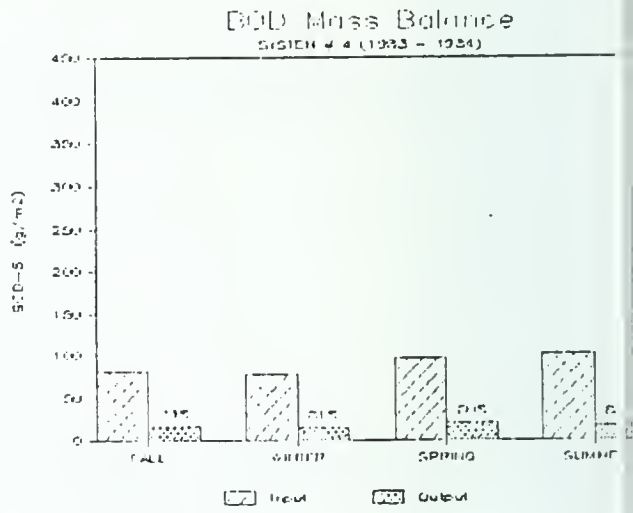

BOD Mass Balance
SYSTEM # 3 (1931 - 1932)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

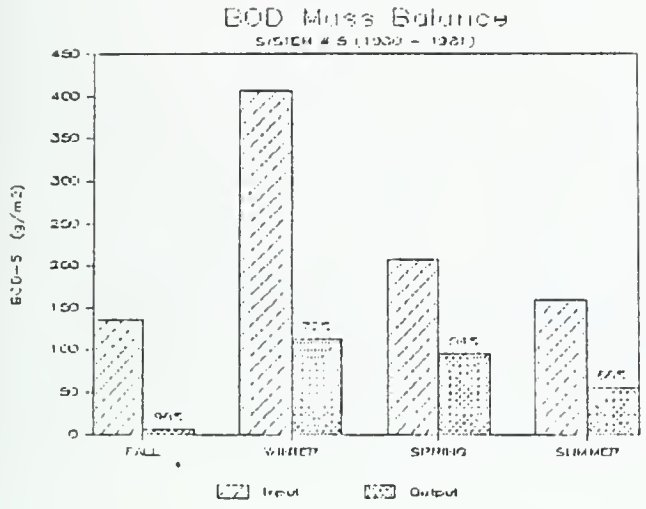

BOD Mass Balance
SYSTEM # 3 (1933 - 1934)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

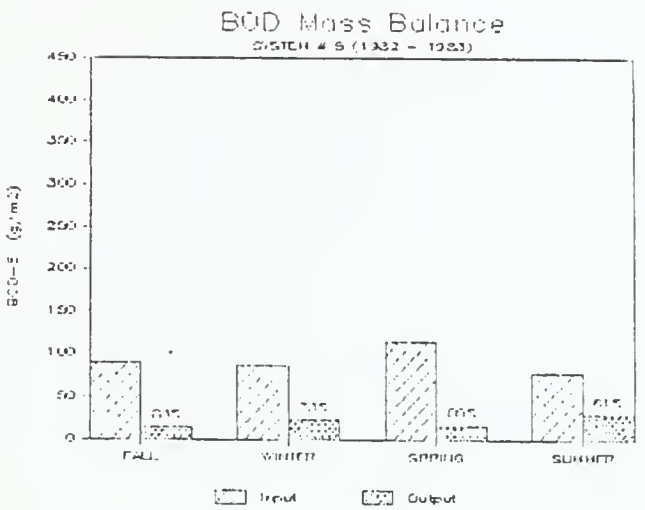

BOD Mass Balance
SISIEN # 4 (1930 - 1931)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

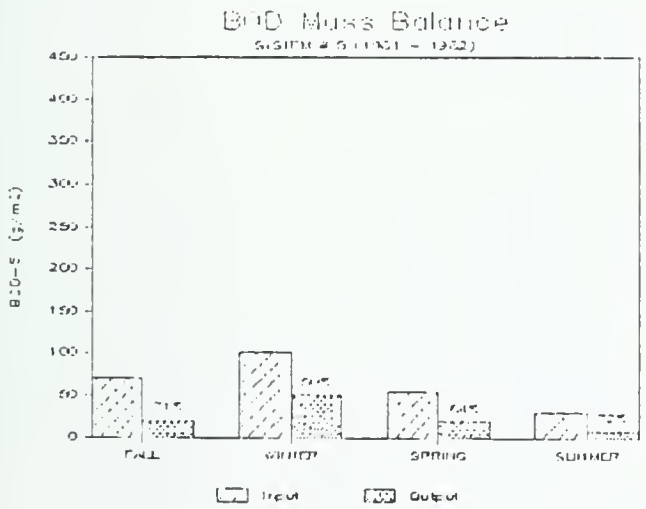

BOD Mass Balance
SISIEN # 4 (1932 - 1933)
BOD-5 (g/m2)
FALL
WINTER
SPRING
Input
Output

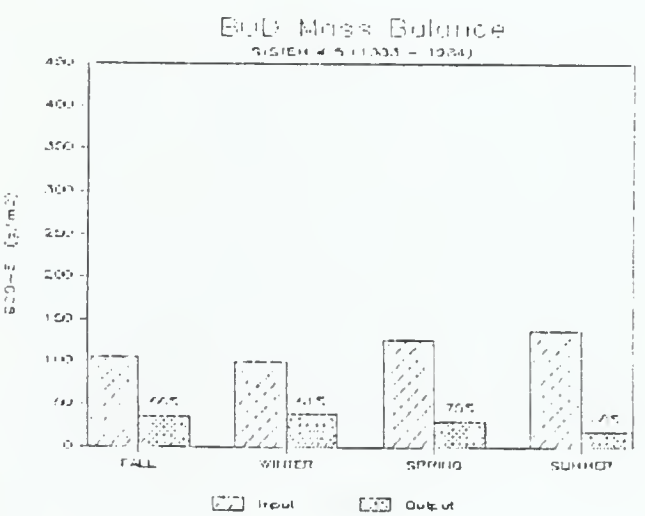

BOD Mass Balance
SISIEN # 4 (1931 - 1932)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

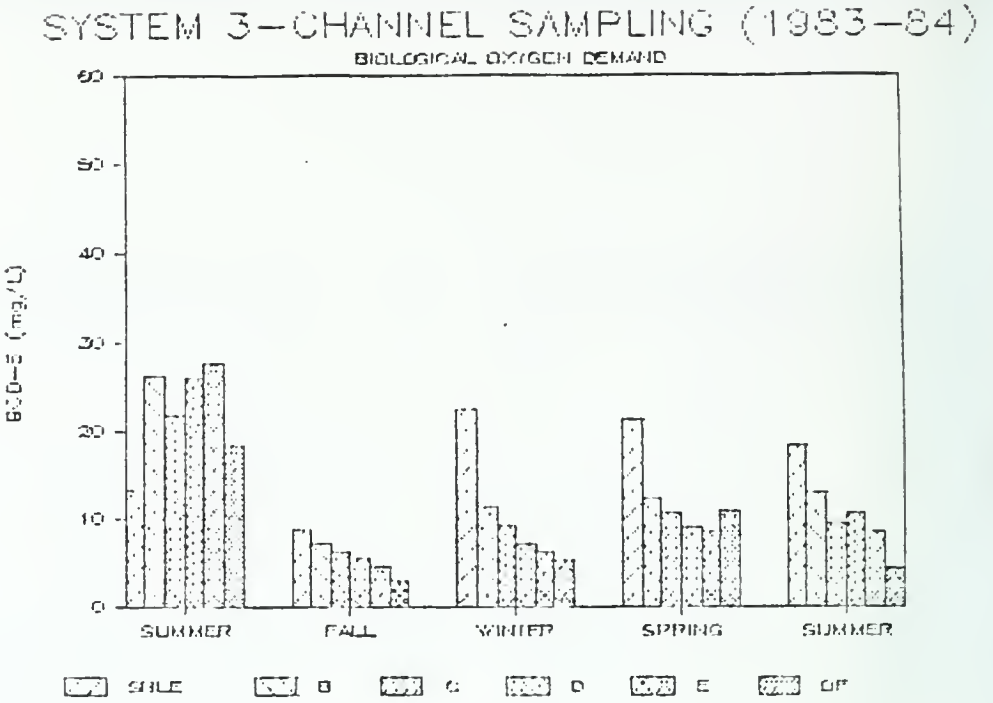

BOD Mass Balance
SISIEN # 4 (1933 - 1934)
BOD-5 (g/m2)
FALL
WINTER
SPRING
Input
Output


BOD Mass Balance
SYSTEM # 5 (1980 - 1981)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


BOD Mass Balance
SYSTEM # 5 (1982 - 1983)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


BOD Mass Balance
SYSTEM # 5 (1981 - 1982)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


BOD Mass Balance
SYSTEM # 5 (1983 - 1984)
BOD-5 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output



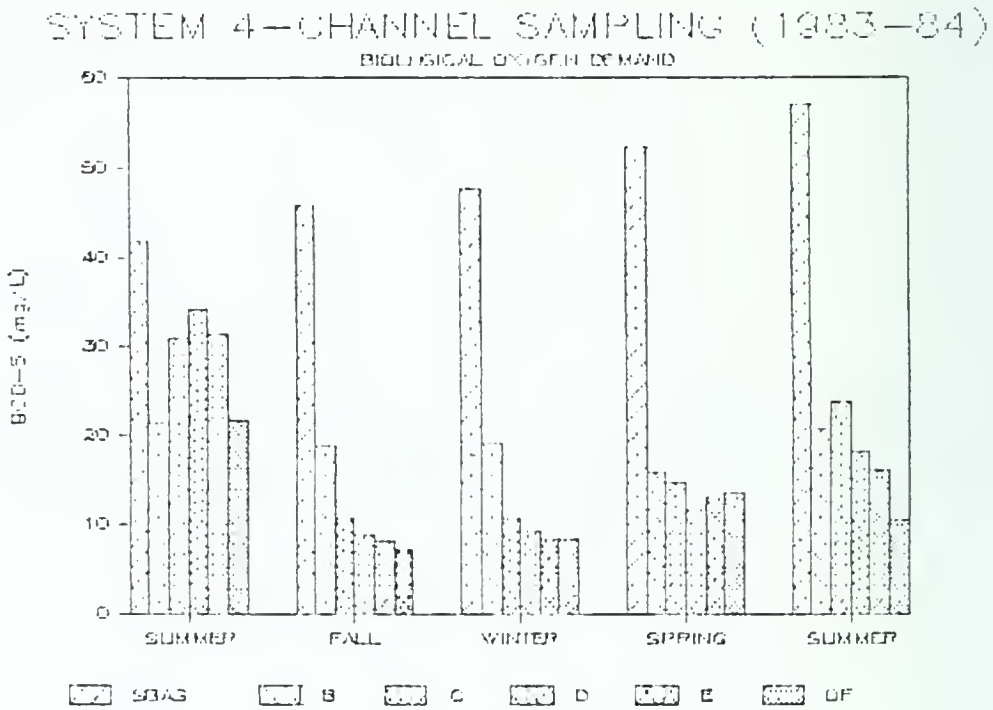


Figure A.6 Seasonal BOD_5 levels in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of Systems 3 and 4 (1983-4).

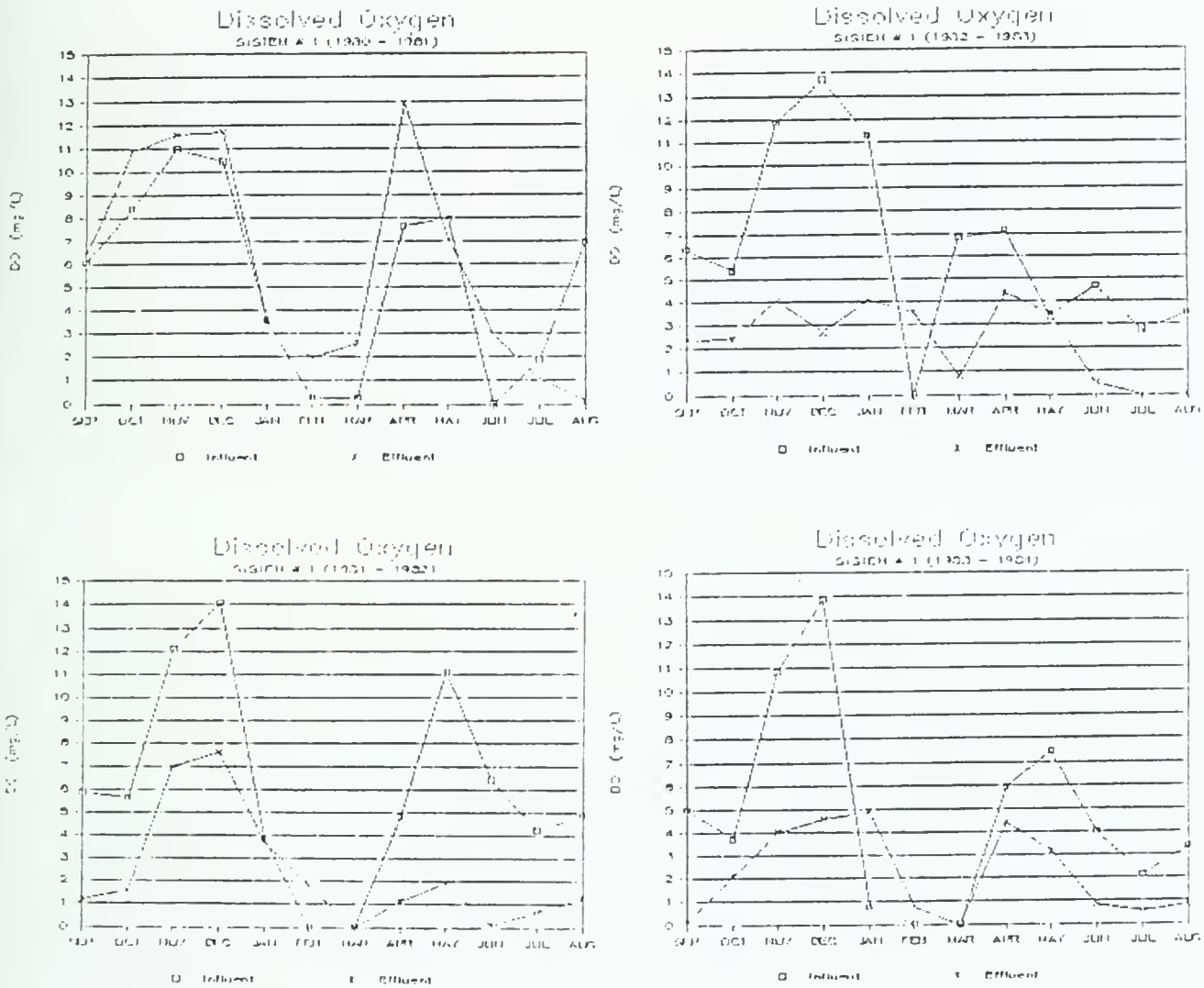
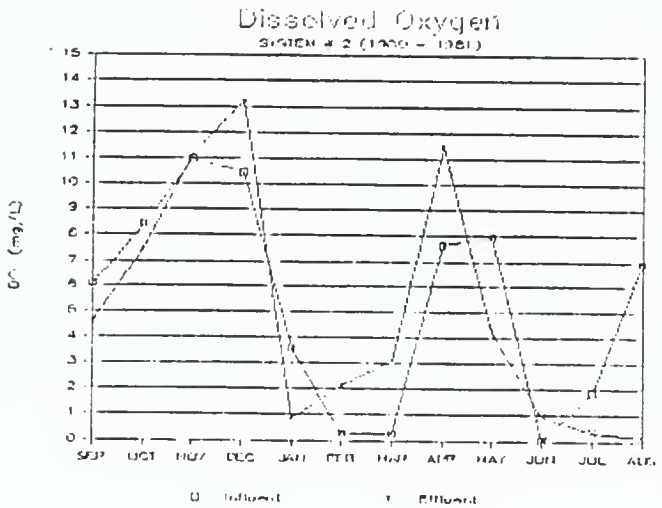
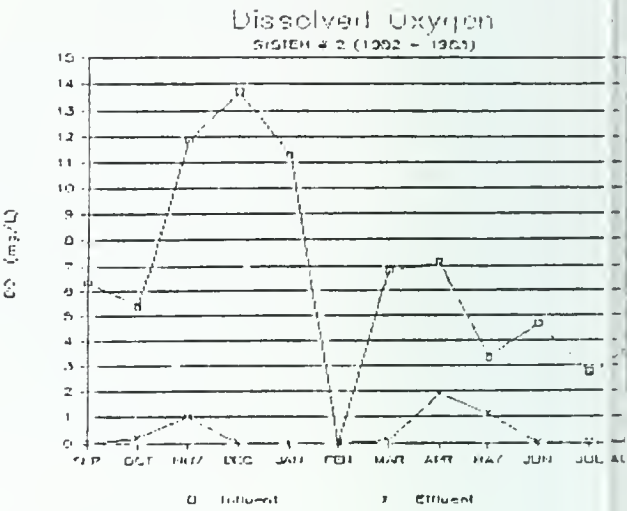
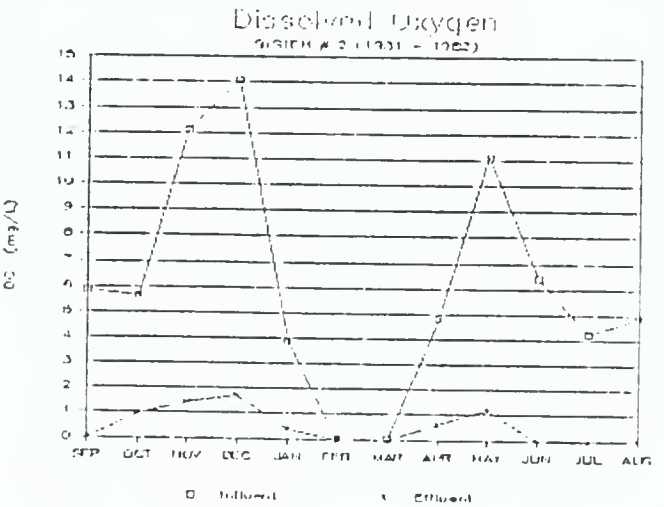
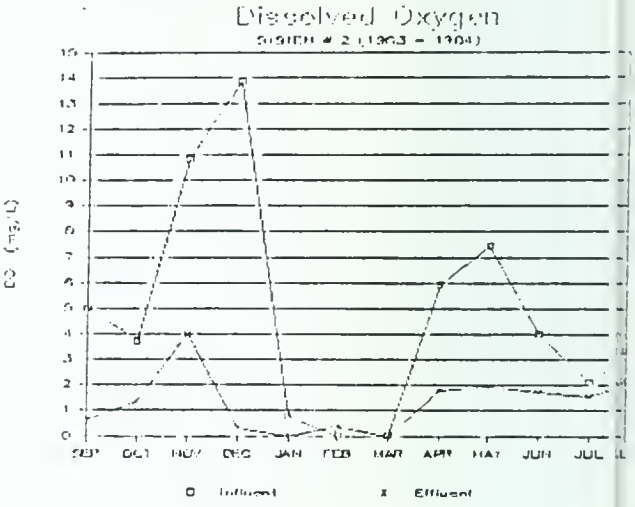


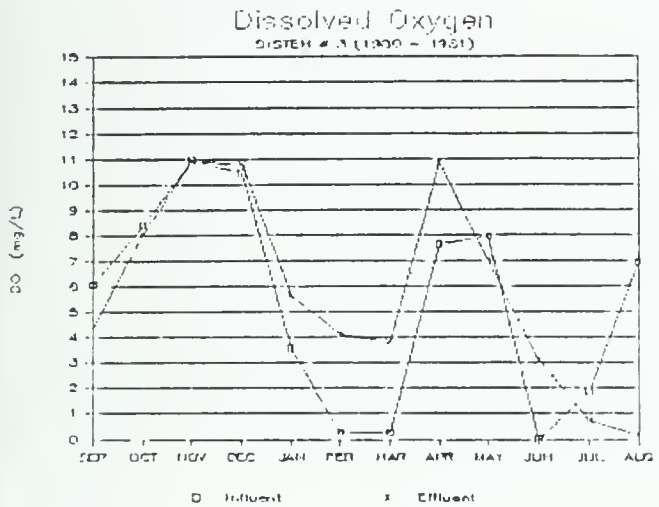
Figure A.7 Monthly average dissolved oxygen concentrations in Systems 1-5 (4 years).

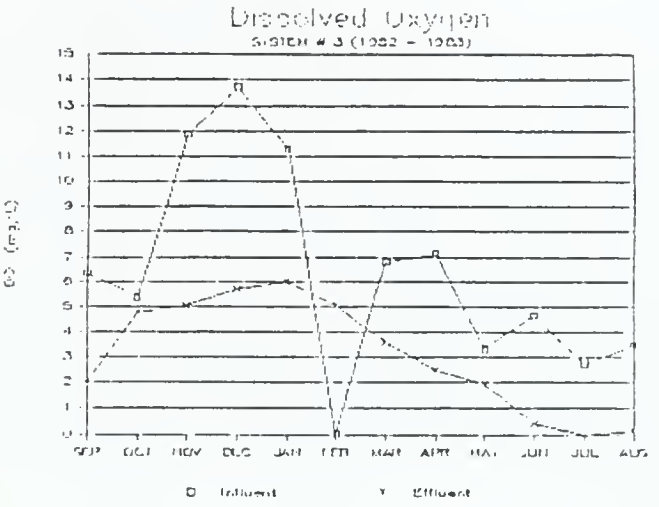

Dissolved Oxygen
SIGIEN # 2 (1980 - 1981)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

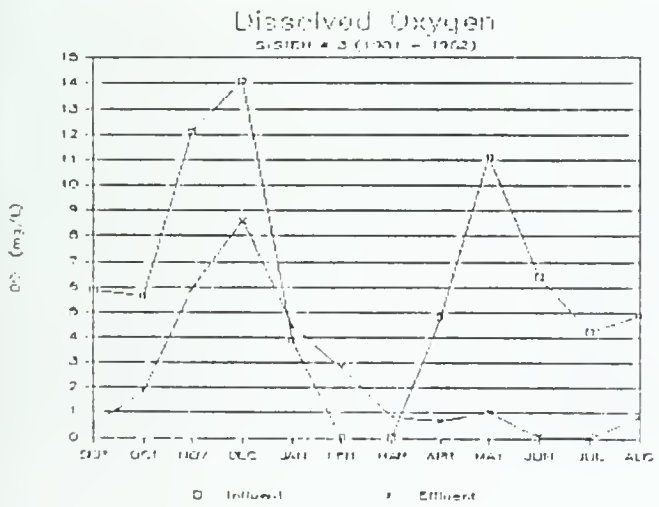

Dissolved Oxygen
SIGIEN # 2 (1982 - 1983)
DO (mg/L)
O Influent
X Effluent

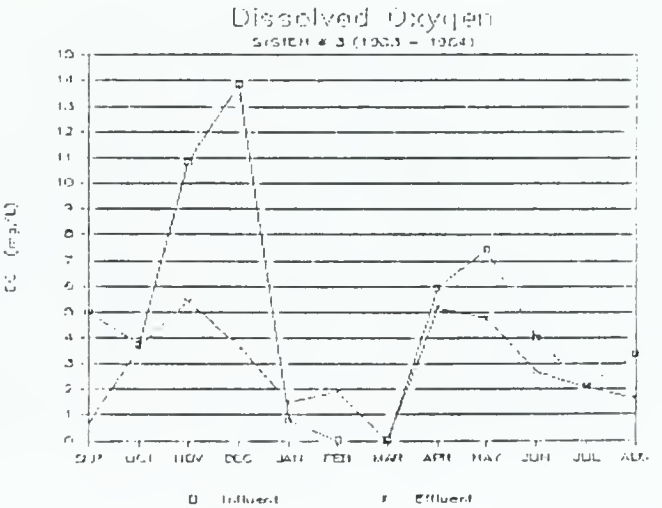

Dissolved Oxygen
SIGIEN # 2 (1981 - 1982)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

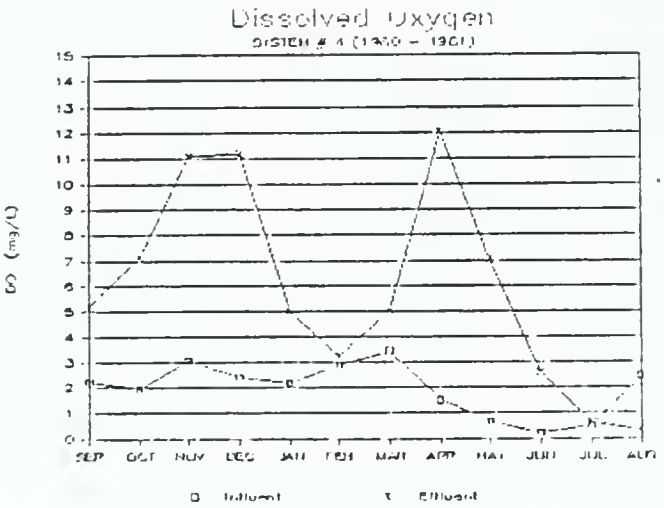

Dissolved Oxygen
SIGIEN # 2 (1983 - 1984)
DO (mg/L)
O Influent
X Effluent

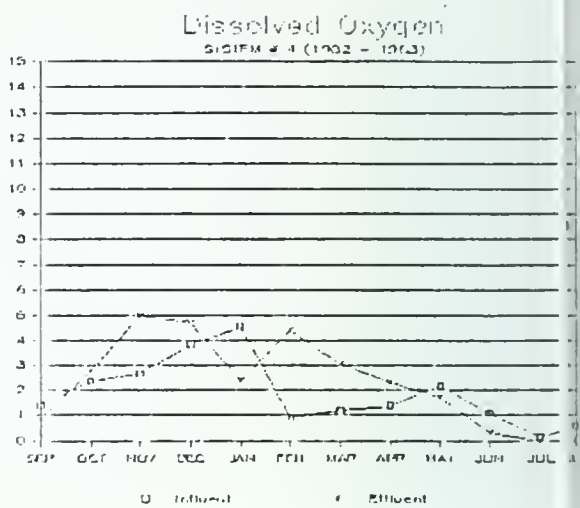

Dissolved Oxygen
SYSTEM # 3 (1980 - 1981)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

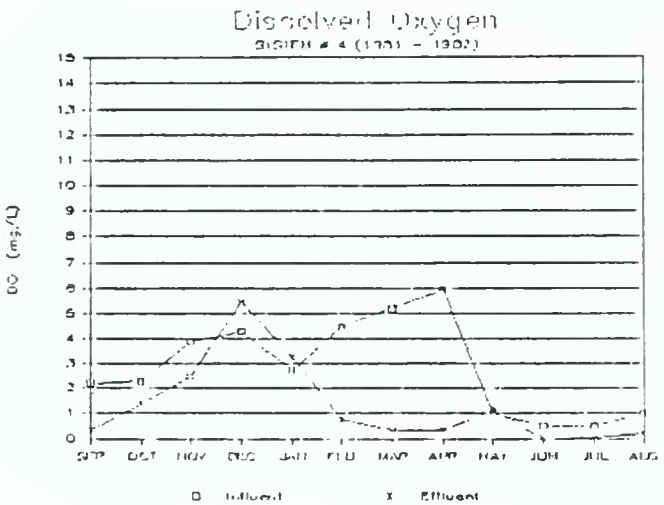

Dissolved Oxygen
SYSTEM # 3 (1982 - 1983)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

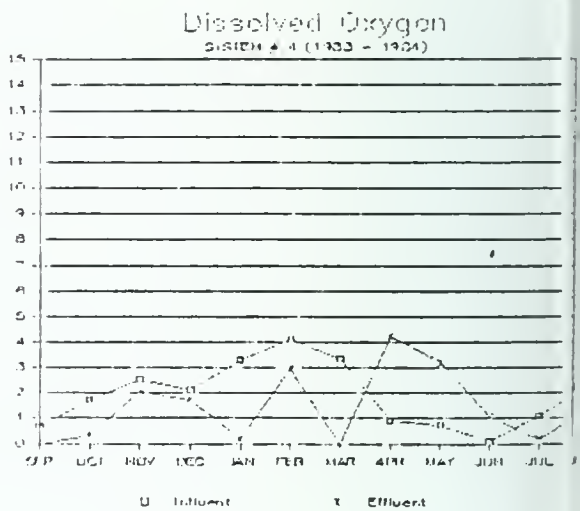

Dissolved Oxygen
SYSTEM # 3 (1981 - 1982)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

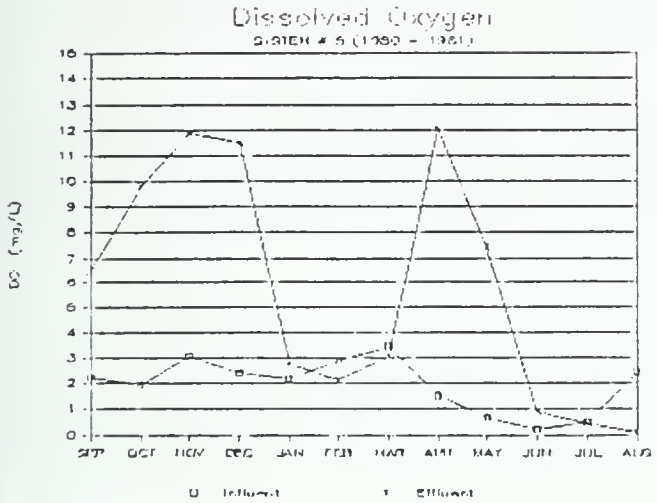

Dissolved Oxygen
SYSTEM # 3 (1983 - 1984)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

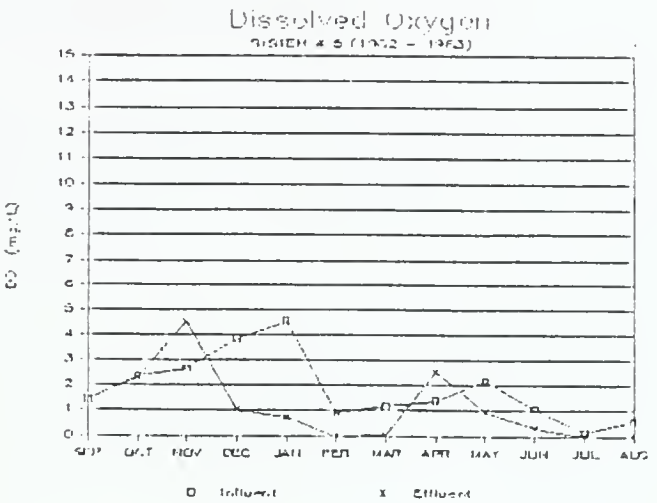

Dissolved Oxygen
SYSTEM # 4 (1900 - 1901)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent X Effluent

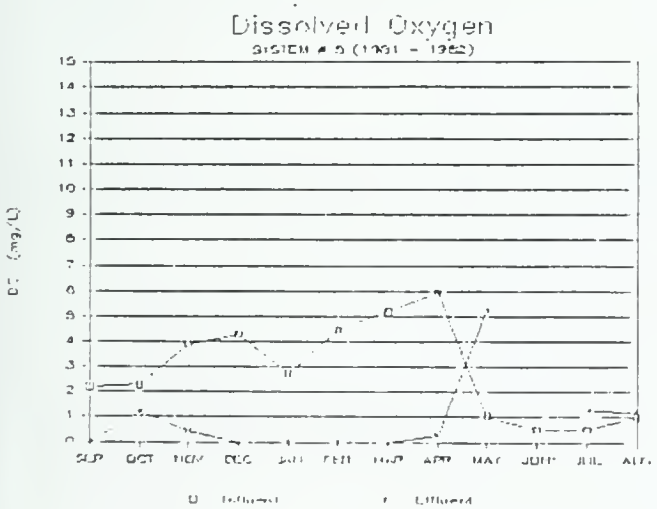

Dissolved Oxygen
SYSTEM # 4 (1902 - 1903)
DO (mg/L)
O Influent X Effluent

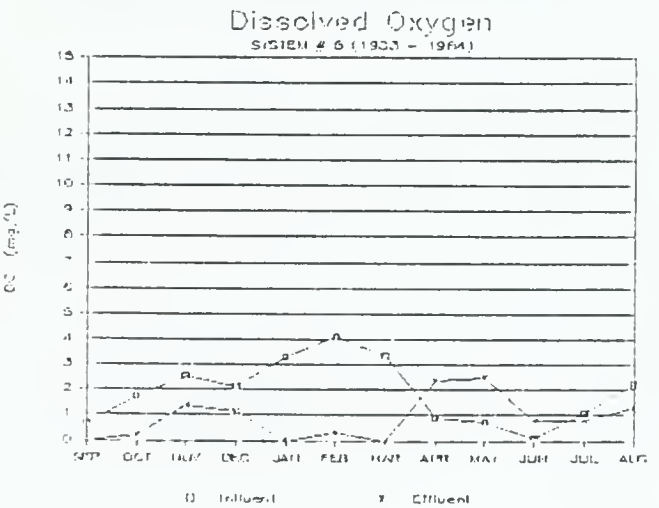

Dissolved Oxygen
SYSTEM # 4 (1901 - 1902)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent X Effluent


Dissolved Oxygen
SYSTEM # 4 (1903 - 1904)
O Influent X Effluent


Dissolved Oxygen
SISTEM # 5 (1960 - 1961)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Dissolved Oxygen
SISTEM # 5 (1962 - 1963)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Dissolved Oxygen
SISTEM # 5 (1961 - 1962)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Dissolved Oxygen
SISTEM # 5 (1963 - 1964)
DO (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

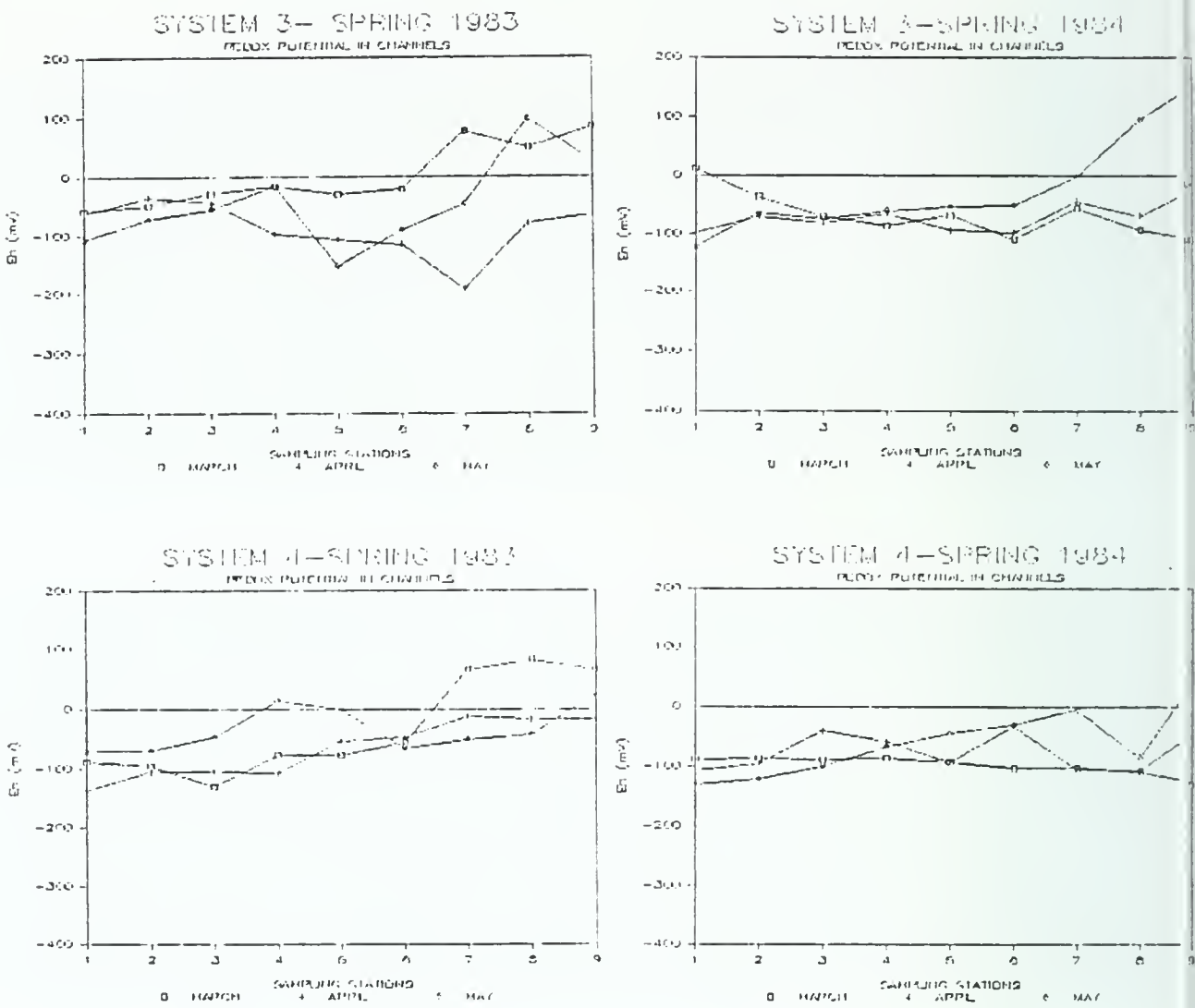
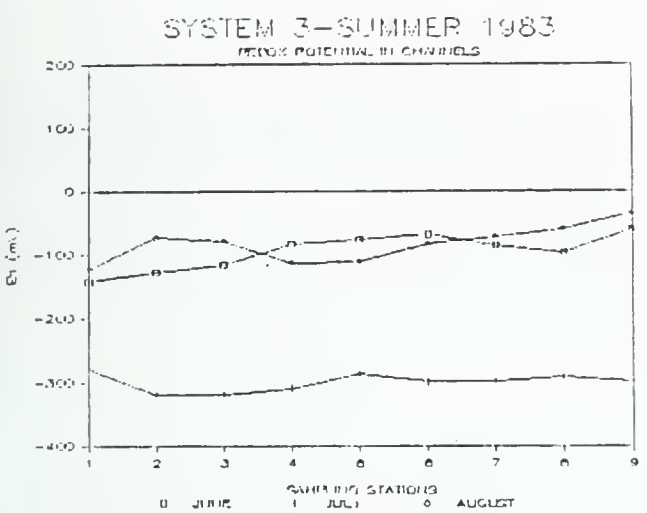
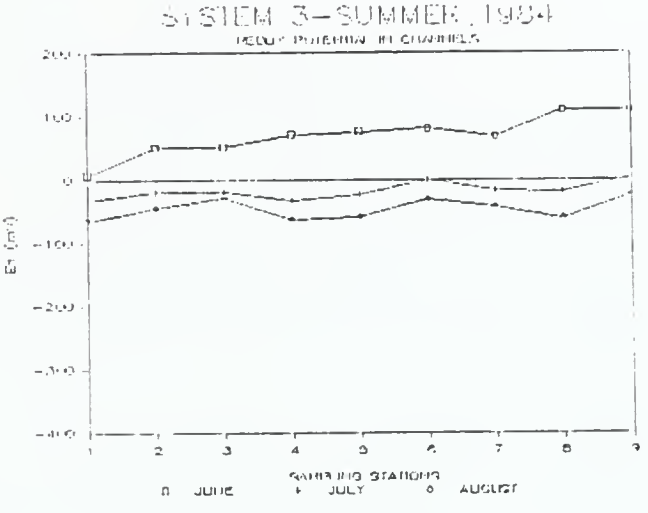
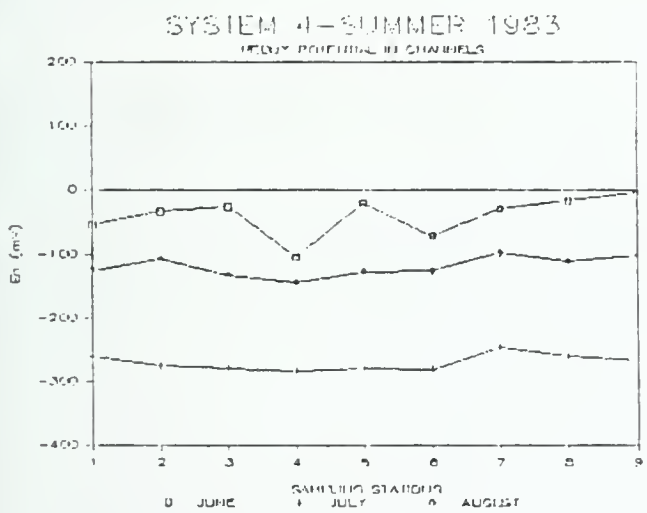
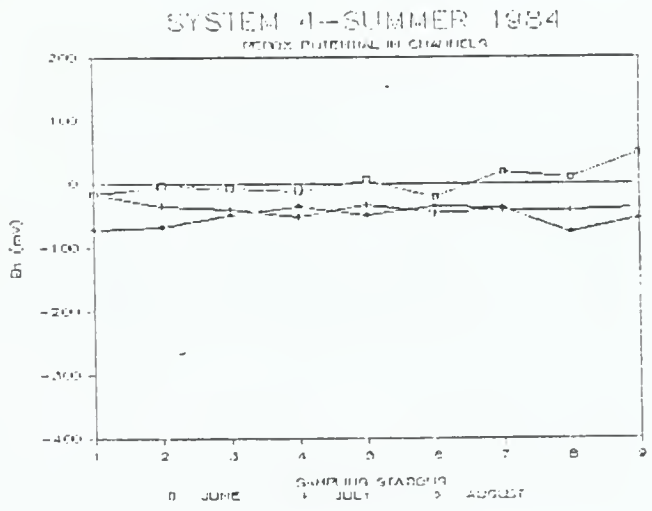


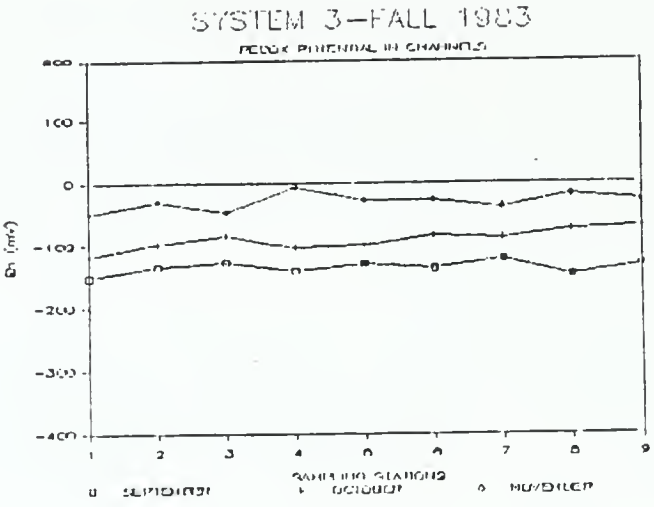
Figure A.8 Monthly average sediment redox potential in channel A (stns. 1-3), C (stns. 4-6) and E (stns. 7-9) of Systems 3 and 4 in spring, summer and fall of 1983 and 1984.

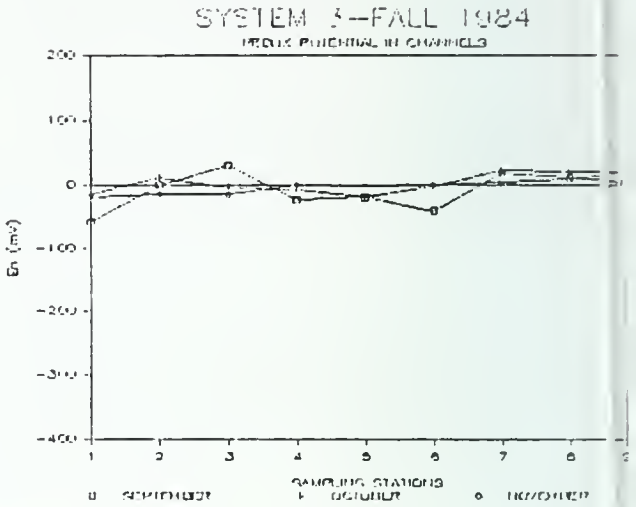

SYSTEM 3—SUMMER 1983
REDOX POTENTIAL IN CHANNELS
Eh (mv)
SAMPLING STATIONS
JUNE
JULY
AUGUST

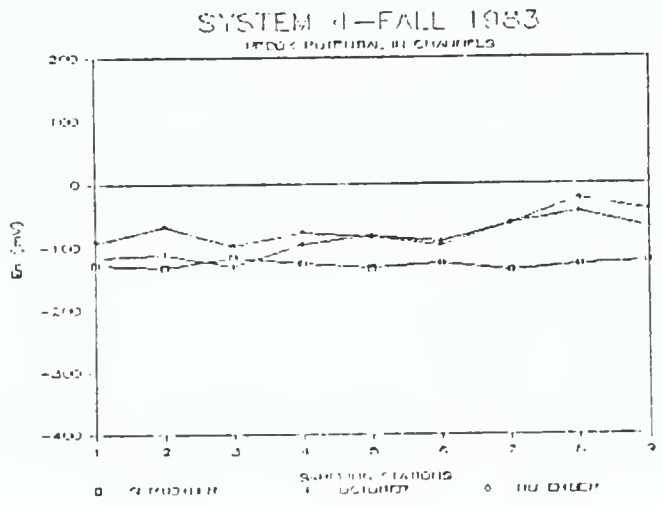

SYSTEM 3—SUMMER 1984
REDOX POTENTIAL IN CHANNELS
Eh (mv)
SAMPLING STATIONS
JUNE
JULY
AUGUST

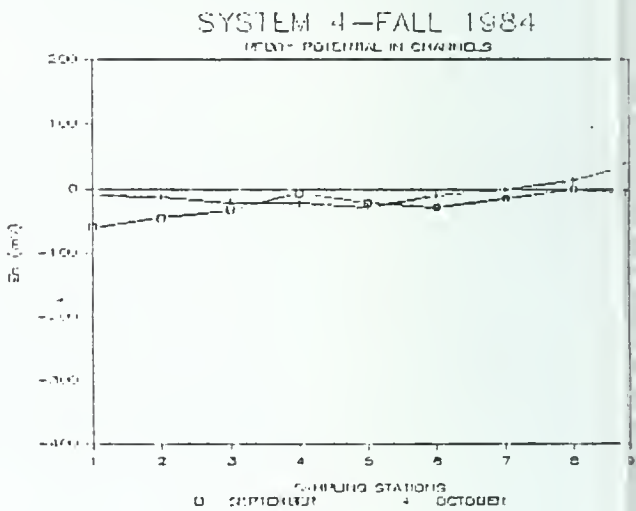

SYSTEM 4—SUMMER 1983
REDOX POTENTIAL IN CHANNELS
Eh (mv)
SAMPLING STATIONS
JUNE
JULY
AUGUST


SYSTEM 4—SUMMER 1984
REDOX POTENTIAL IN CHANNELS
Eh (mv)
SAMPLING STATIONS
JUNE
JULY
AUGUST


SYSTEM 3—FALL 1983
REDOX POTENTIAL IN CHANNELS
100
0
-100
-200
-300
-400
1
2
3
4
5
6
7
8
9
SAMPLING STATIONS
OCTOBER
NOVEMBER


SYSTEM 3—FALL 1984
REDOX POTENTIAL IN CHANNELS
200
100
0
-100
-200
-300
-400
1
2
3
4
5
6
7
8
SAMPLING STATIONS
OCTOBER
NOVEMBER


SYSTEM 4—FALL 1983
REDOX POTENTIAL IN CHANNELS
200
100
0
-100
-200
-300
-400
1
2
3
4
5
6
7
8
9
SAMPLING STATIONS
OCTOBER


SYSTEM 4—FALL 1984
REDOX POTENTIAL IN CHANNELS
200
100
0
-100
-300
-400
1
2
3
4
5
6
7
8
SAMPLING STATIONS
OCTOBER

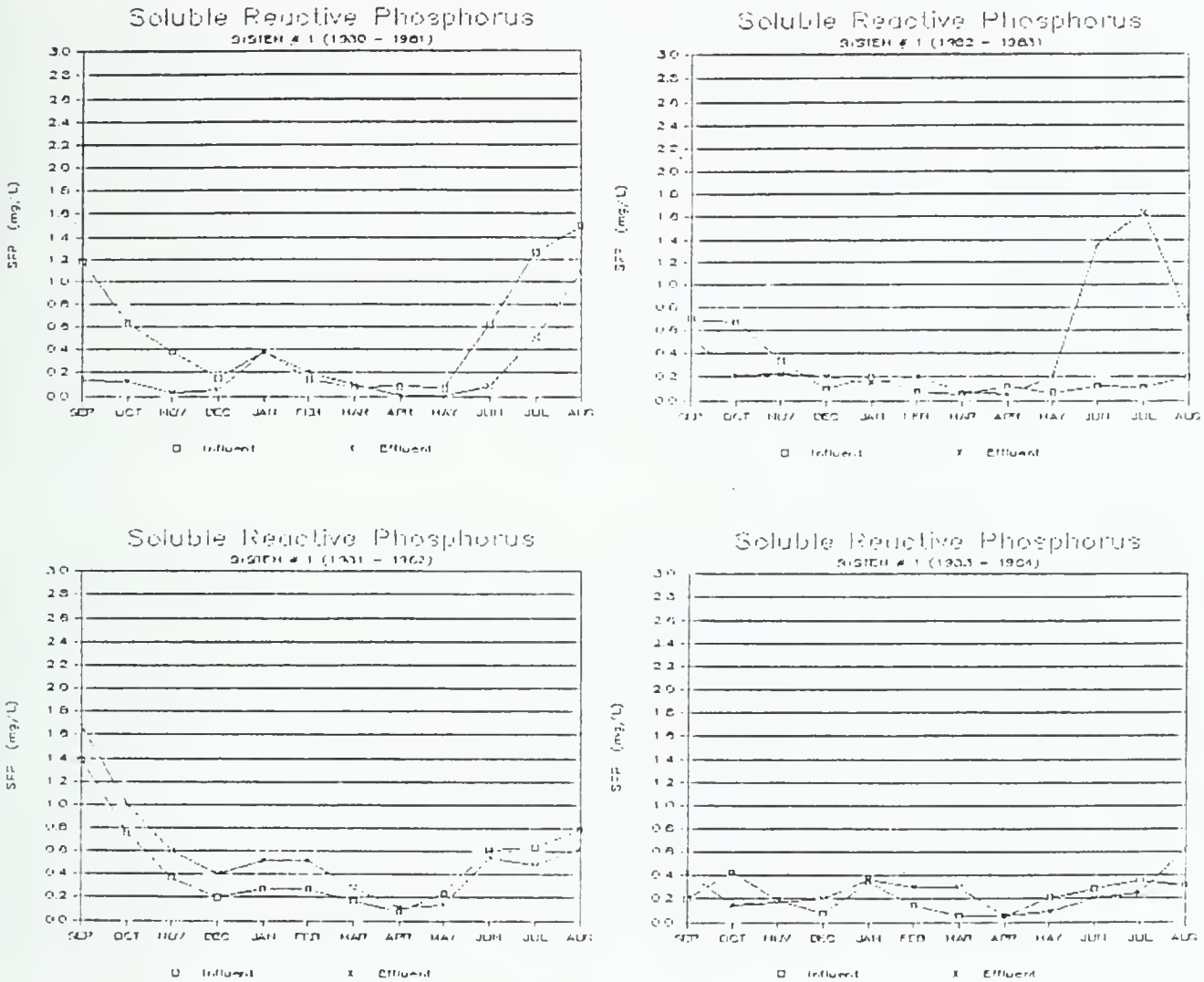
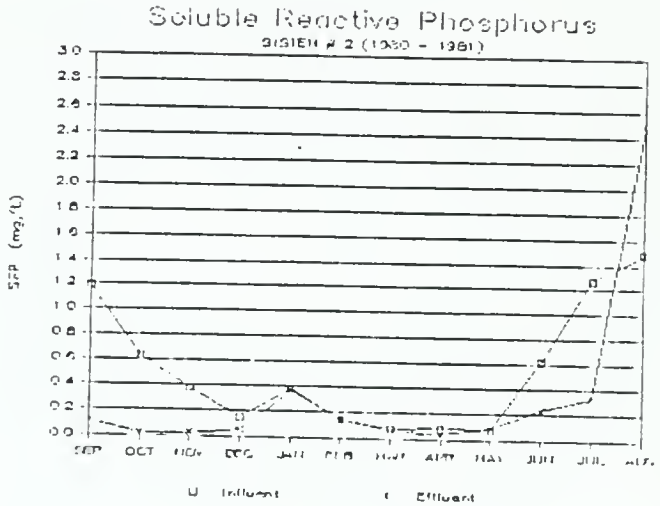
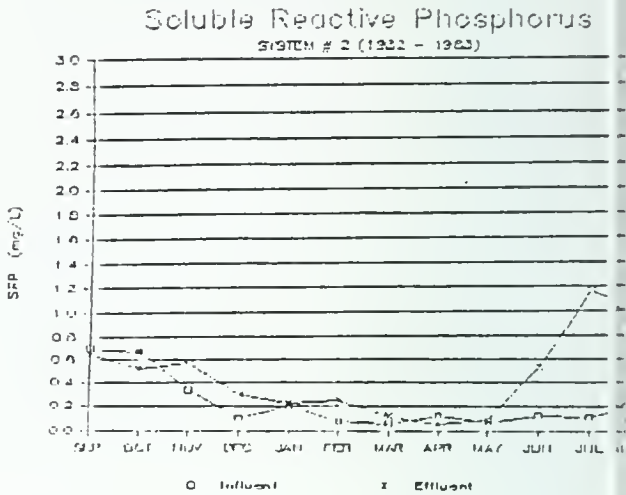
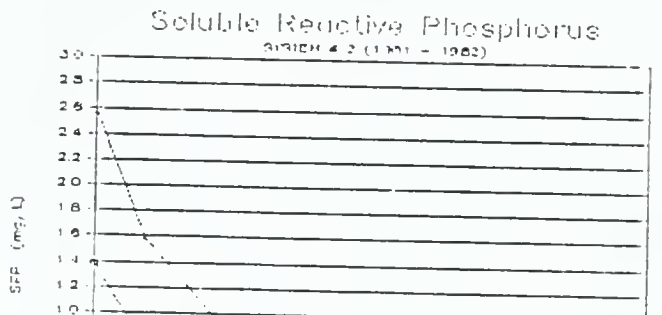
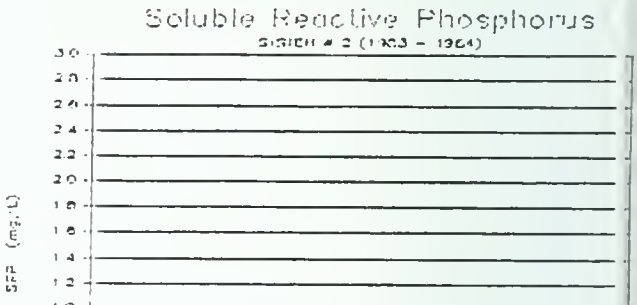


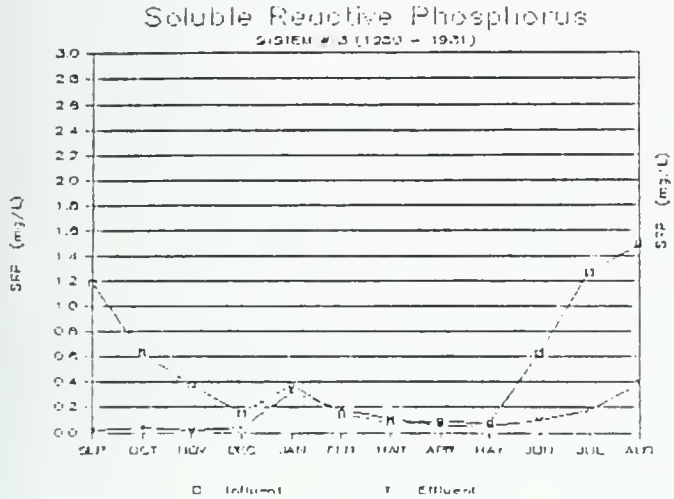
Figure A.9 Monthly average soluble phosphorus concentrations in Systems 1-5 (4 years).

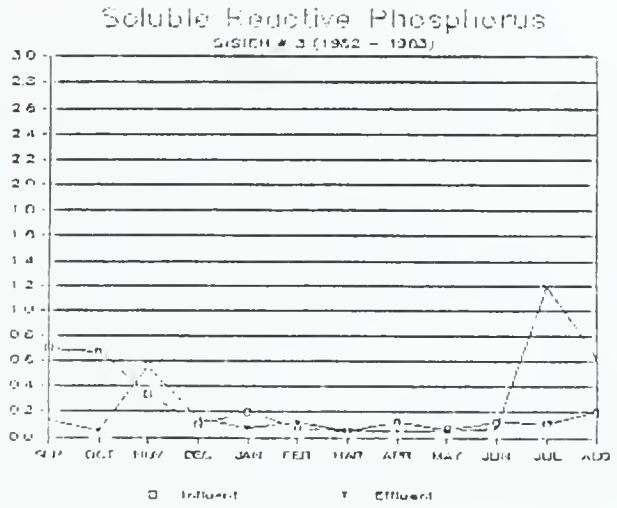

Soluble Reactive Phosphorus
SYSTEM # 2 (1980 - 1981)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

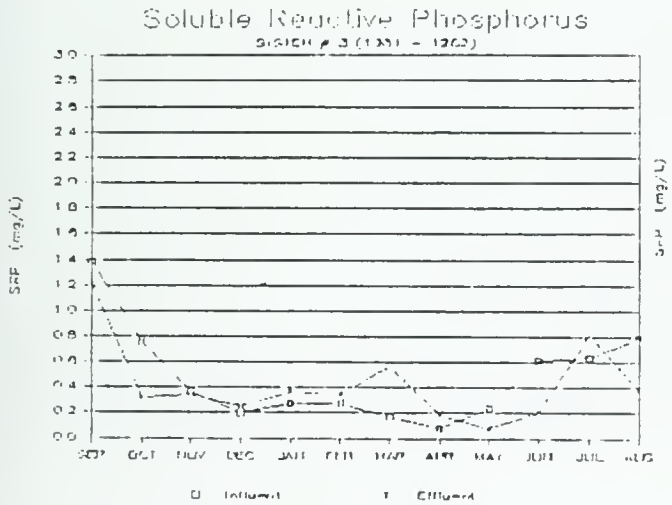

Soluble Reactive Phosphorus
SYSTEM # 2 (1982 - 1983)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

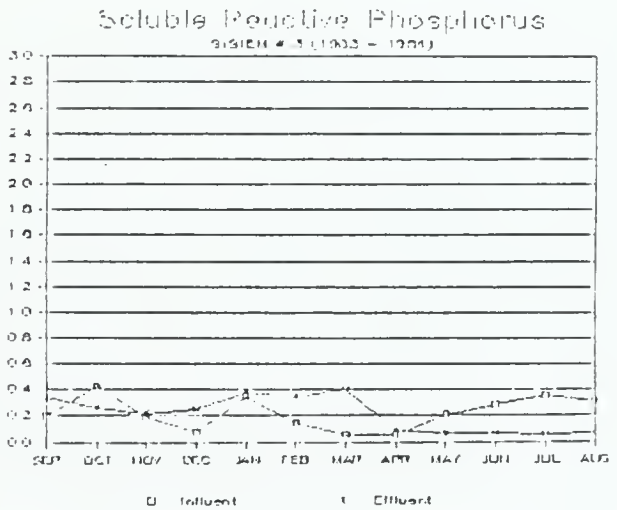

Soluble Reactive Phosphorus
SYSTEM # 2 (1981 - 1982)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

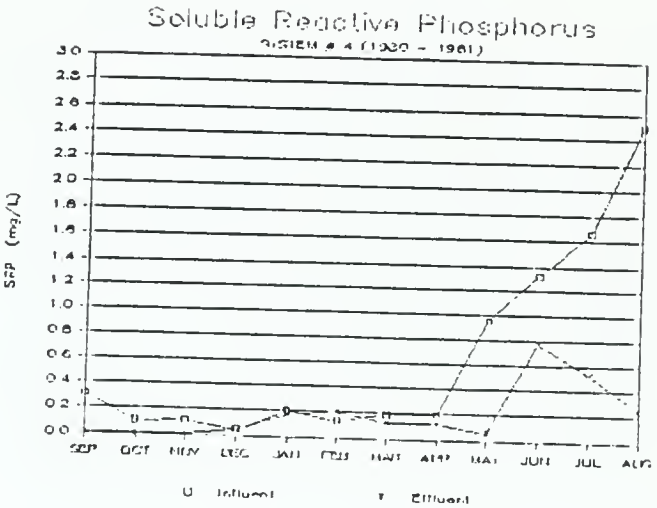

Soluble Reactive Phosphorus
SYSTEM # 2 (1983 - 1984)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

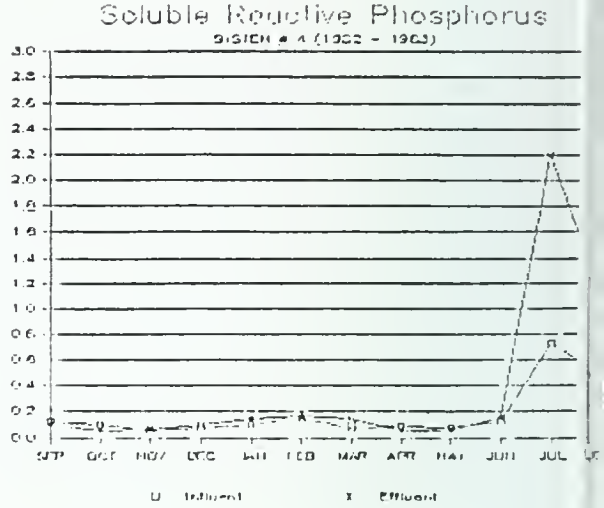

Soluble Reactive Phosphorus
SRP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

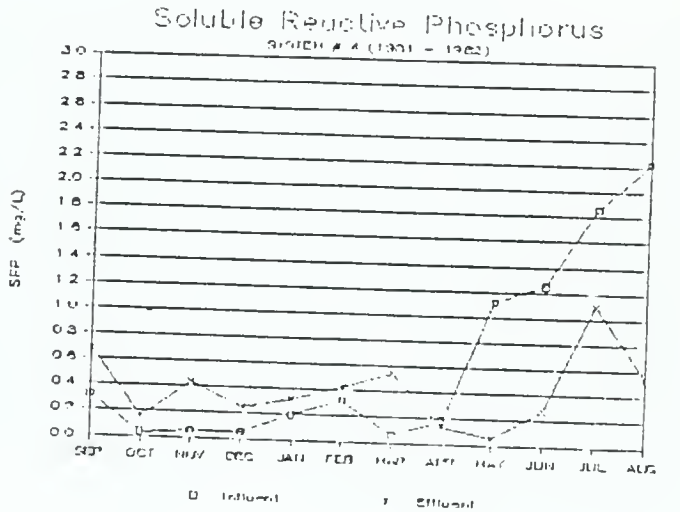

Soluble Reactive Phosphorus
SRP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

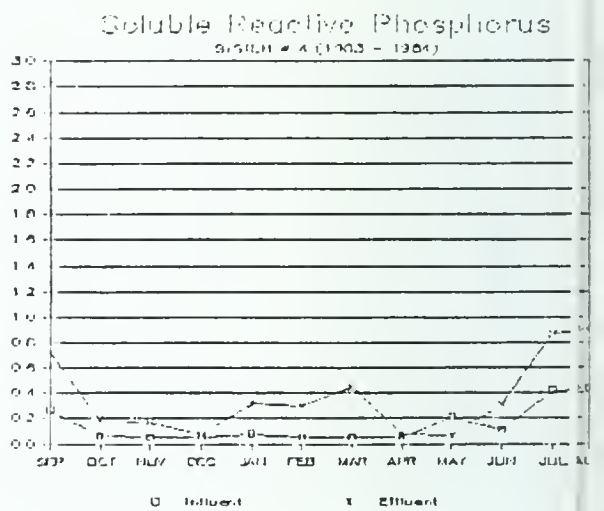

Soluble Reactive Phosphorus
SRP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

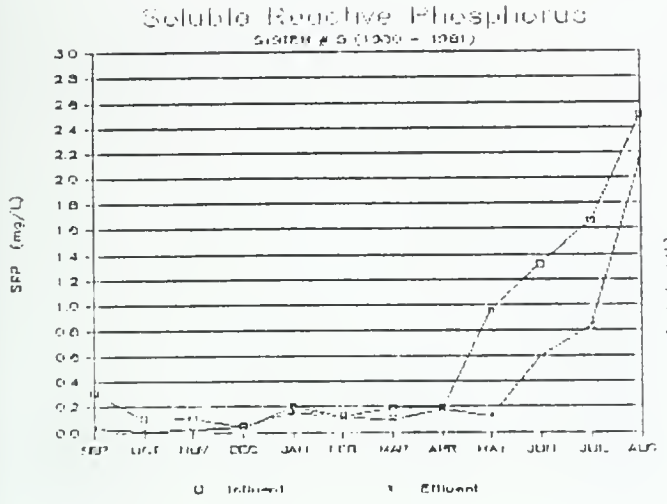

Soluble Reactive Phosphorus
SRP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

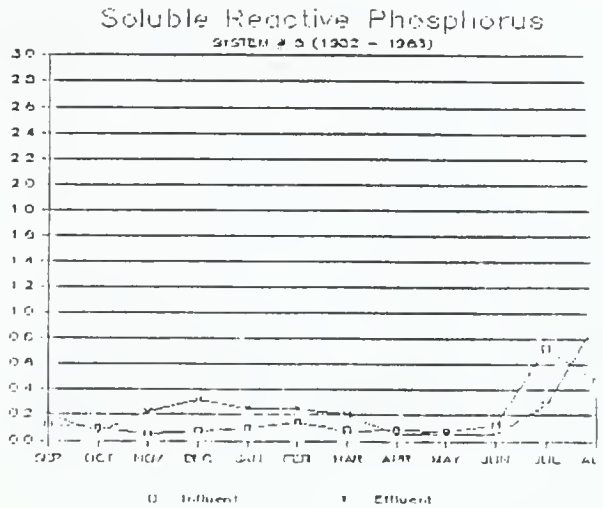

Soluble Reactive Phosphorus
SISIEM # 4 (1980 - 1981)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

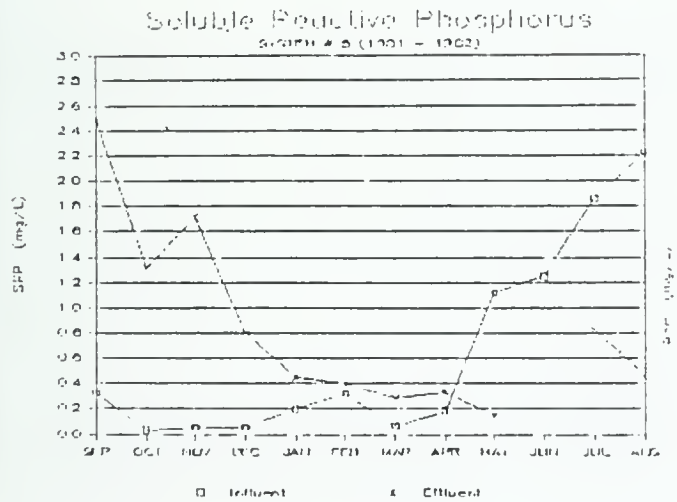

Soluble Reactive Phosphorus
SISIEM # 4 (1982 - 1983)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

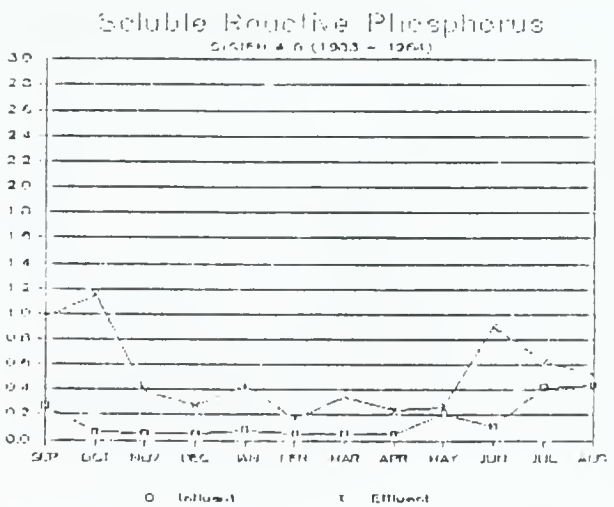

Soluble Reactive Phosphorus
SISIEM # 4 (1981 - 1982)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Soluble Reactive Phosphorus
SISIEM # 4 (1983 - 1984)
SRP (mg/L)
3.0
2.8
2.6
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent


Soluble Reactive Phosphorus
SRP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Soluble Reactive Phosphorus
SRP (mg/L)
O Influent
X Effluent


Soluble Reactive Phosphorus
SRP (mg/L)
O Influent
X Effluent


Soluble Reactive Phosphorus
O Influent
X Effluent

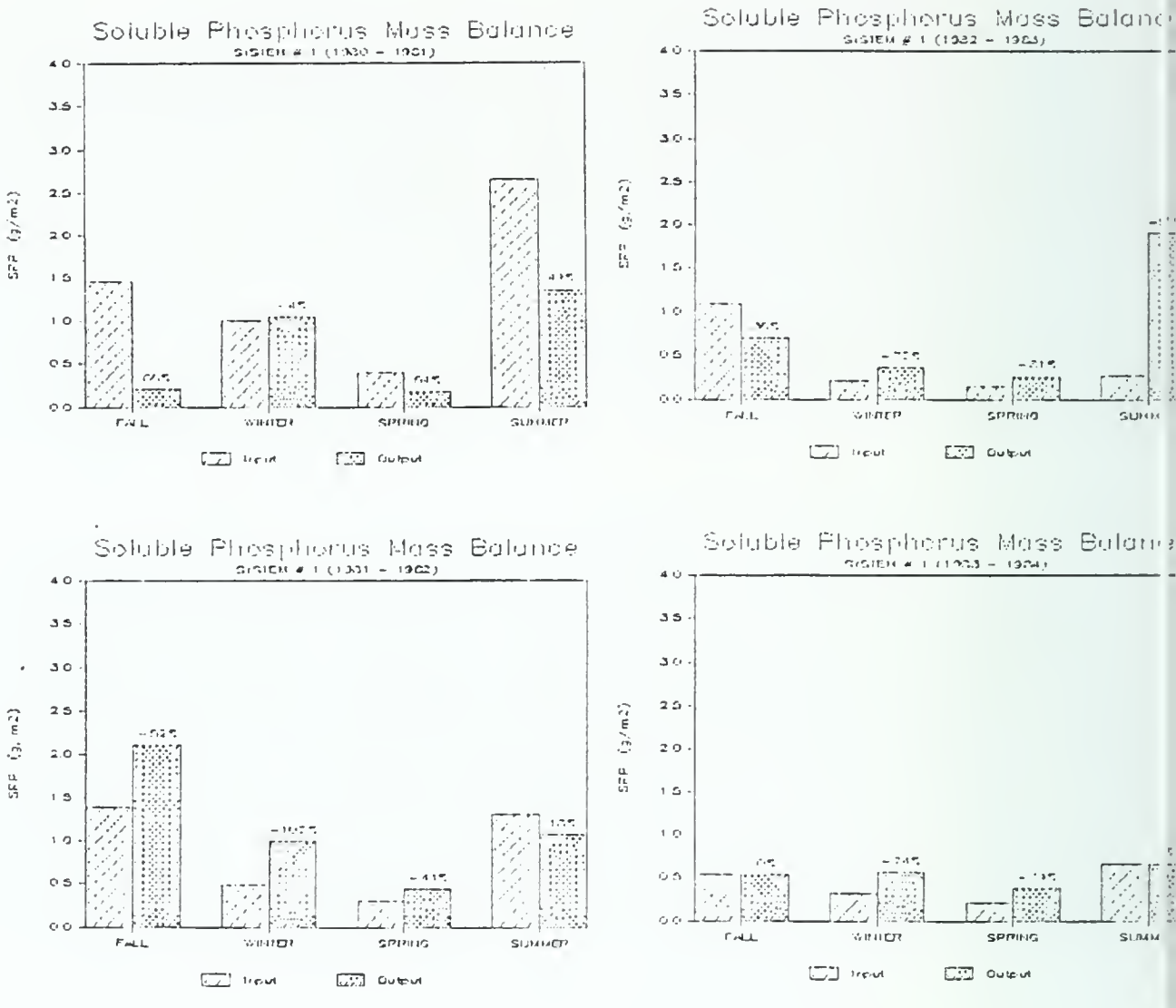
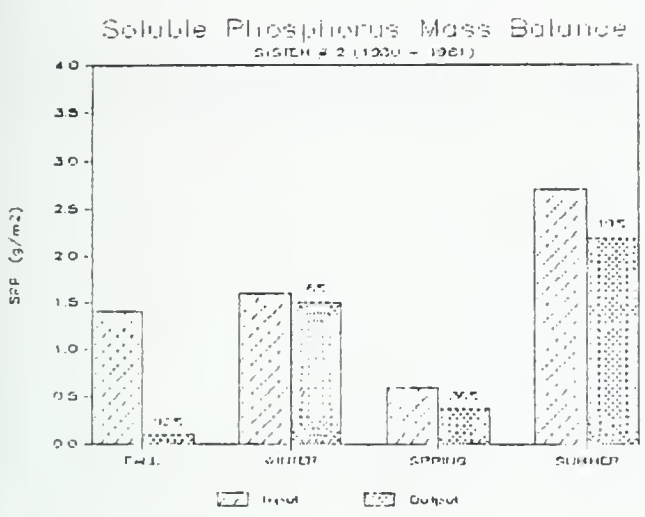
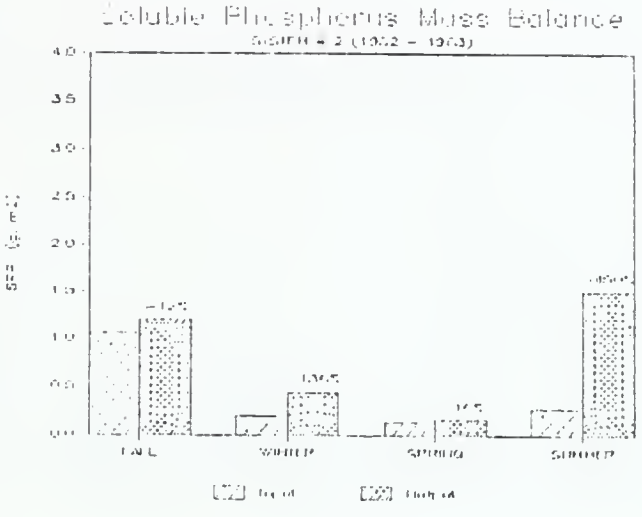
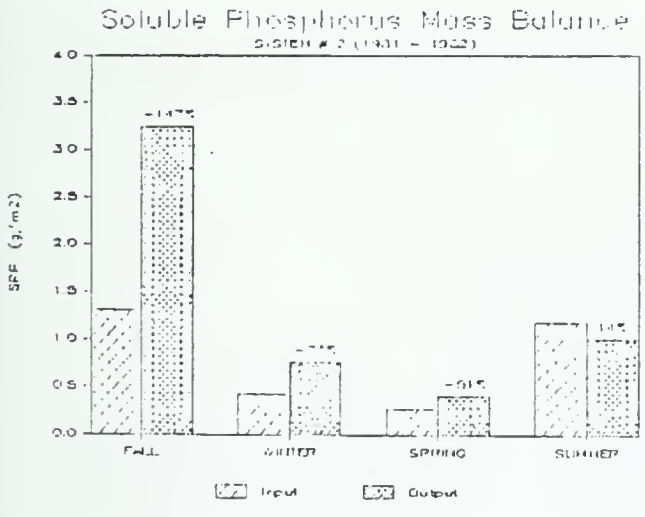
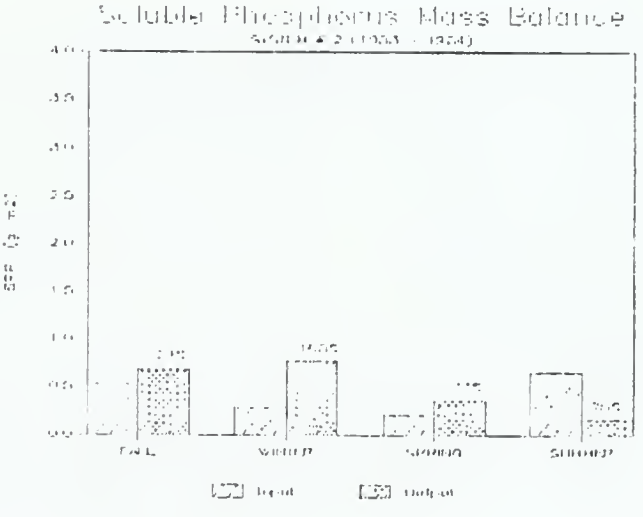


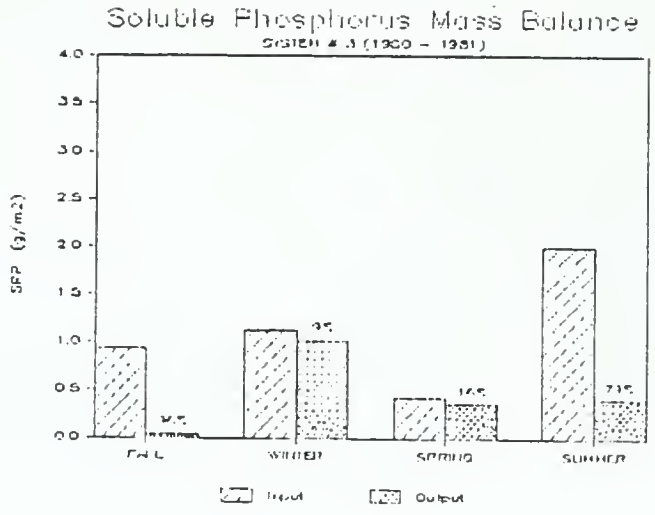
Figure A.10 Seasonal soluble phosphorus mass balances and removal efficiencies in Systems 1-5 (4 years).

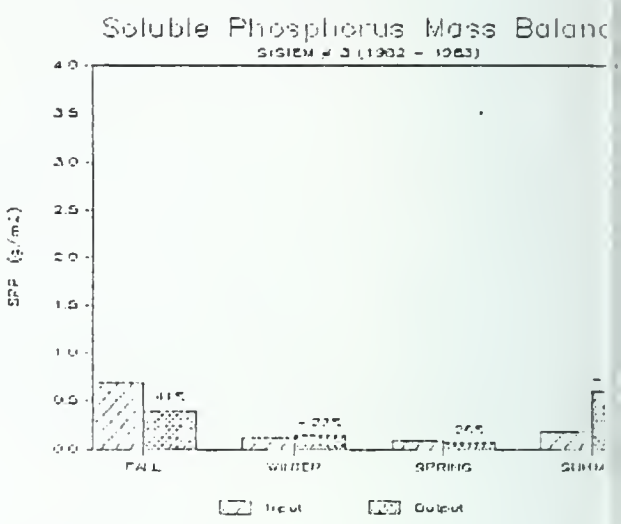

Soluble Phosphorus Mass Balance
SISTEN # 2 (1980 - 1981)
SPP (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

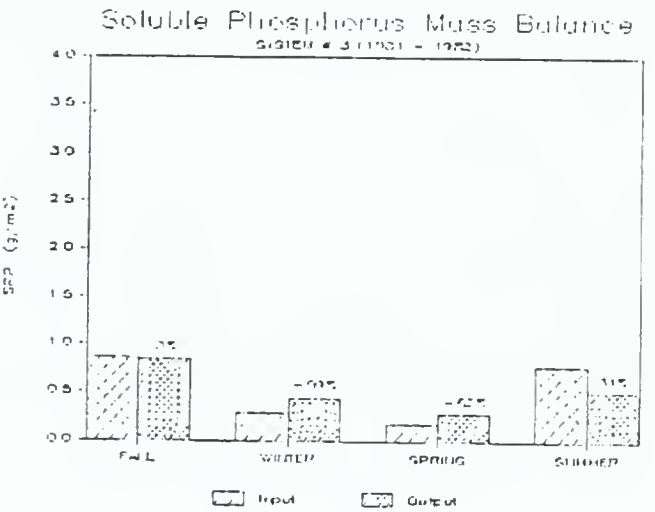

Soluble Phosphorus Mass Balance
SISTEN # 2 (1982 - 1983)
SPP (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

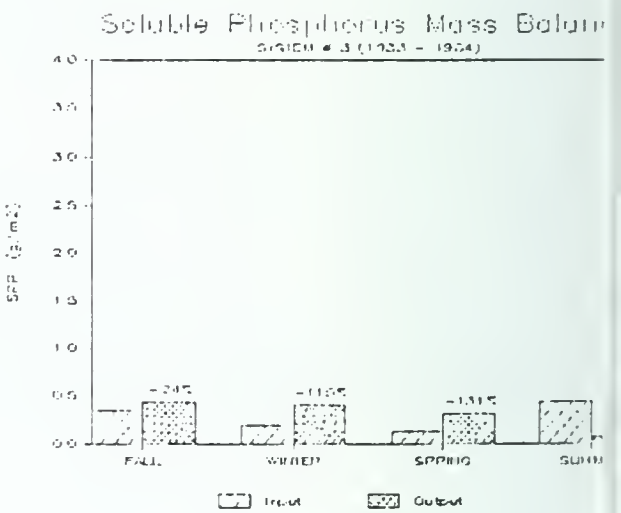

Soluble Phosphorus Mass Balance
SISTEN # 2 (1981 - 1982)
SPP (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

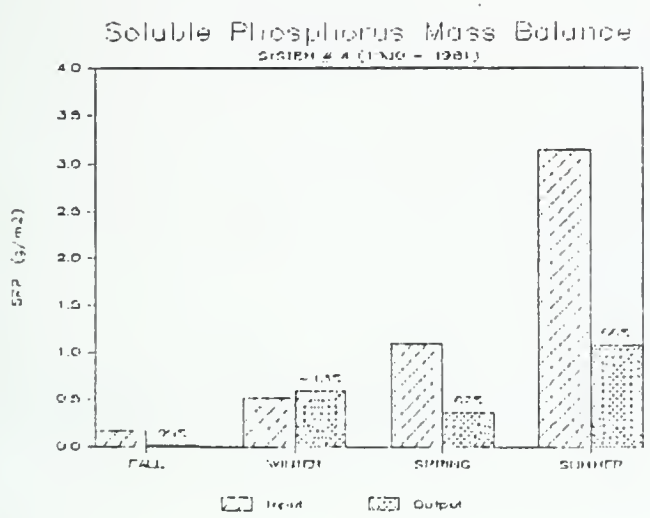

Soluble Phosphorus Mass Balance
SPP (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

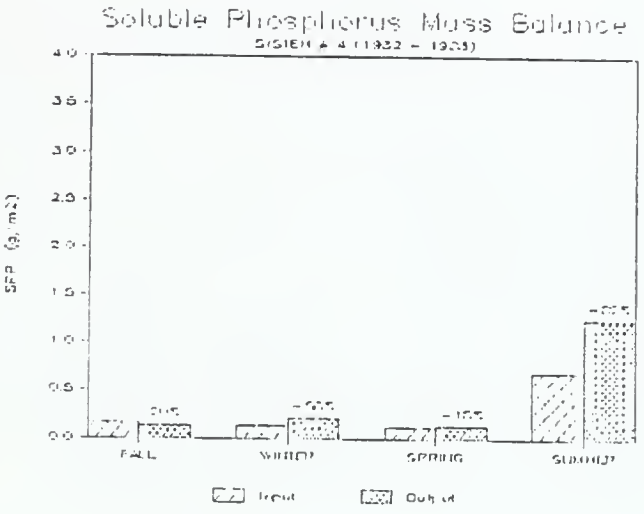

Soluble Phosphorus Mass Balance
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

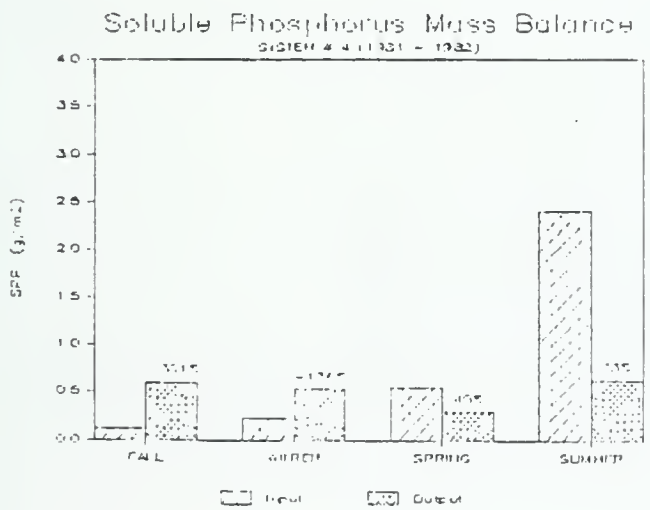

Soluble Phosphorus Mass Balanc
SRP (g/m2)
FALL
WINTER
SPRING
Input
Output

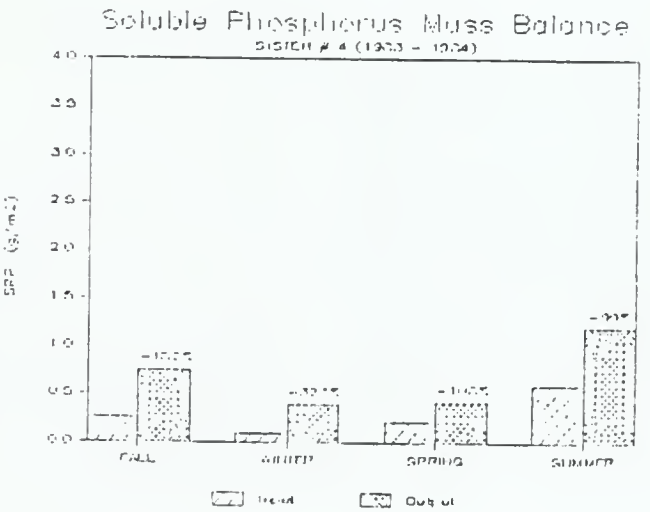

Soluble Phosphorus Mass Balance
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

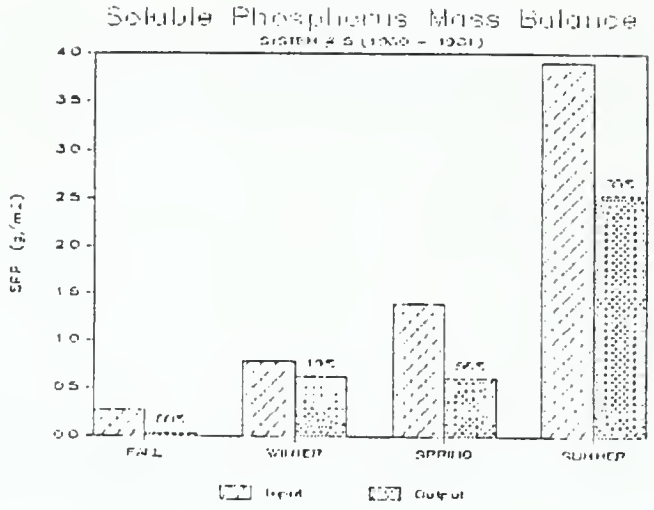

Soluble Phosphorus Mass Balan
SRP (g/m2)
FALL
WINTER
SPRING
Input
Output

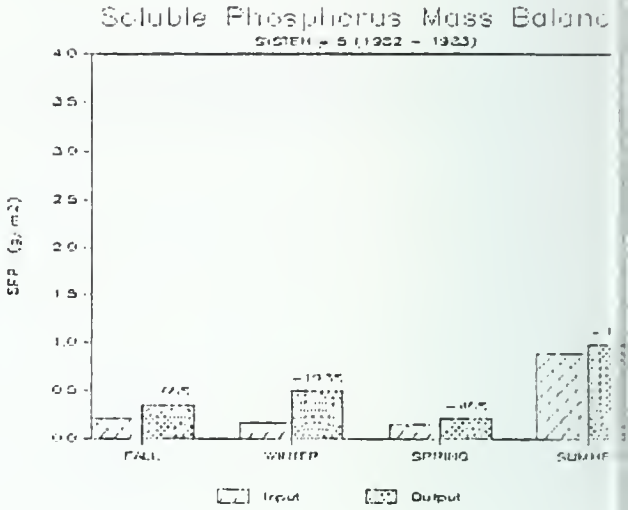

Soluble Phosphorus Mass Balance
SISTEN # 4 (1930 - 1981)
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

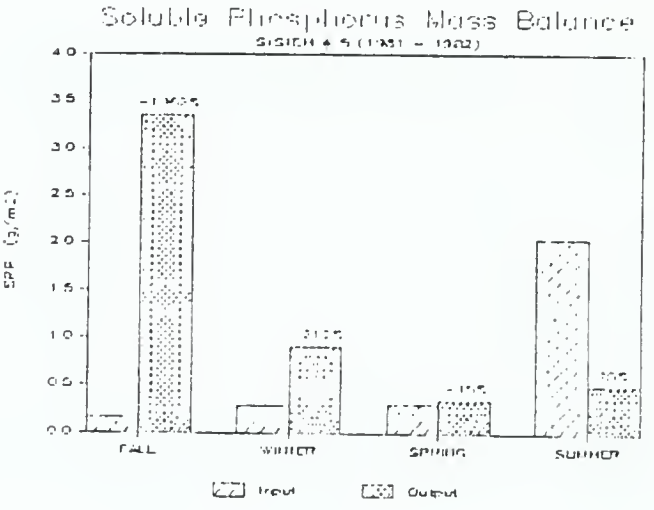

Soluble Phosphorus Mass Balance
SISTEN # 4 (1932 - 1933)
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

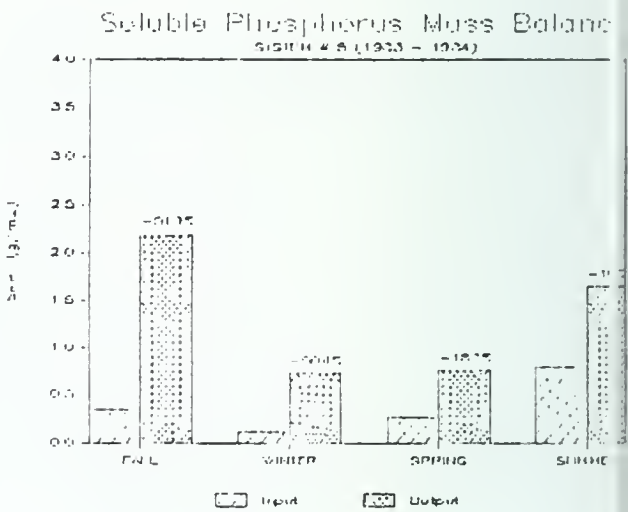

Soluble Phosphorus Mass Balance
SISTEN # 4 (1931 - 1932)
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Soluble Phosphorus Mass Balance
SISTEN # 4 (1933 - 1934)
SRP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Soluble Phosphorus Mass Balance
SISTEN # 5 (1930 - 1931)
SPP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Soluble Phosphorus Mass Balanc
SISTEN # 5 (1932 - 1933)
SPP (g/m2)
FALL
WINTER
SPRING
Input
Output


Soluble Phosphorus Mass Balance
SISTEN # 5 (1931 - 1932)
SPP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Soluble Phosphorus Mass Balanc
SISTEN # 5 (1933 - 1934)
FALL
WINTER
SPRING
Input
Output

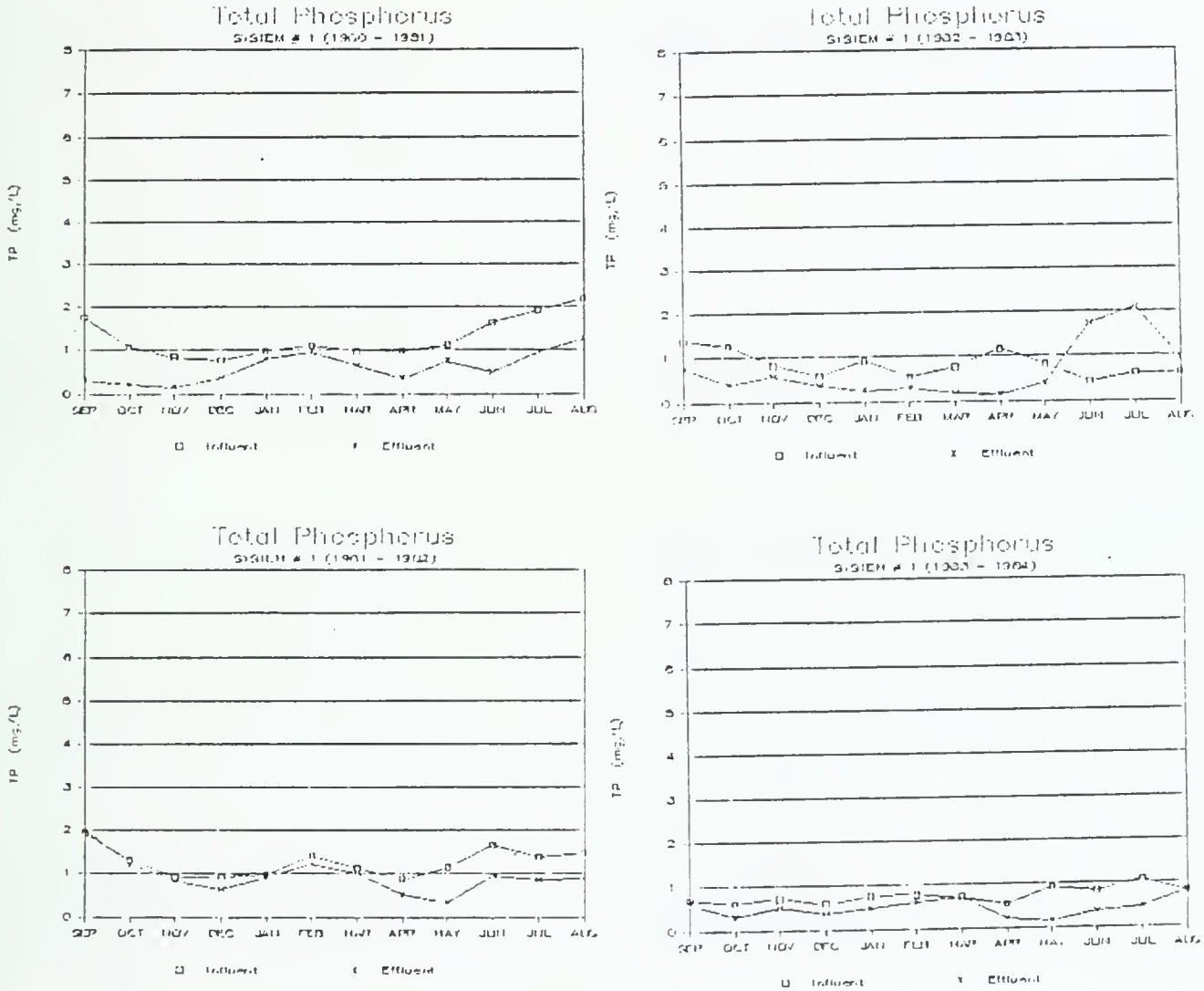
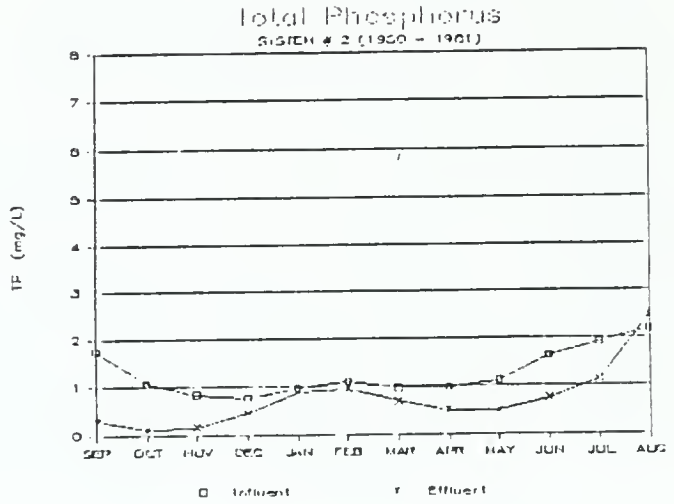
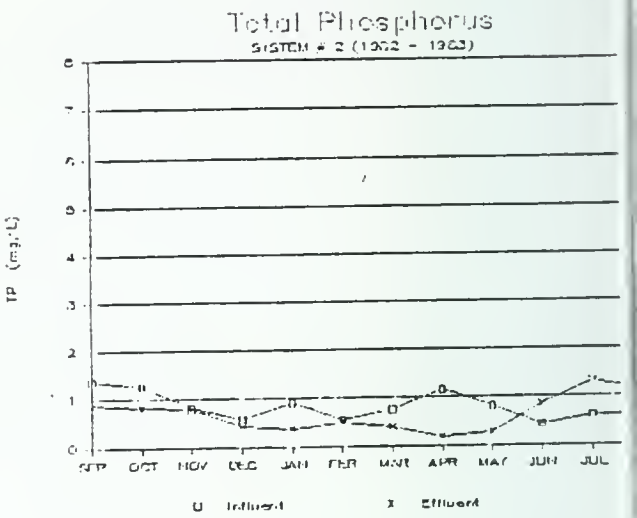
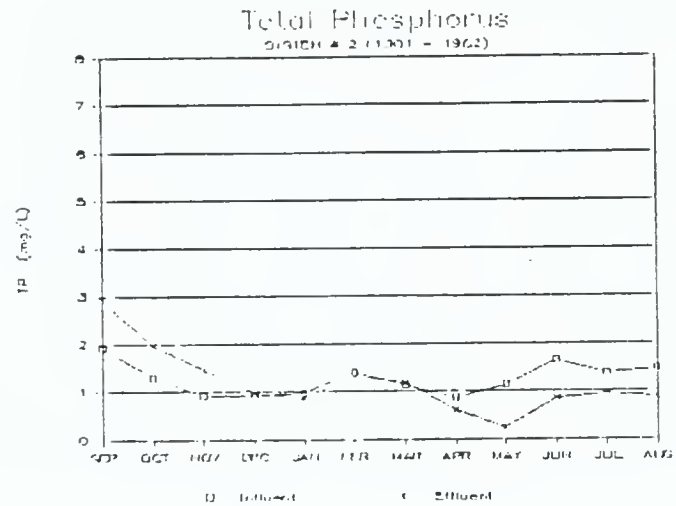
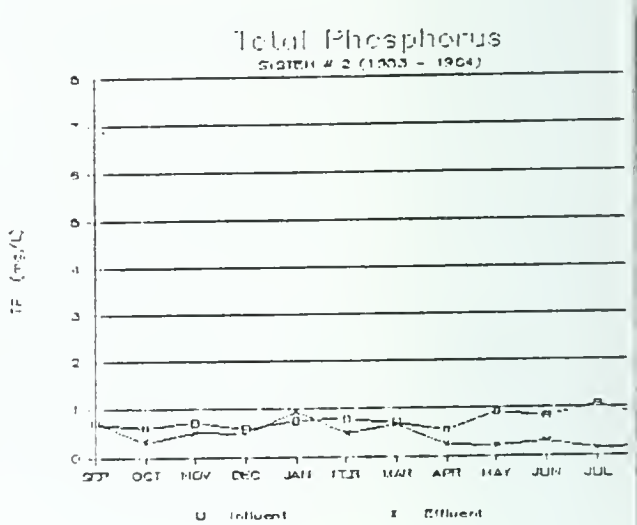

Total Phosphorus
SYSTEM # 1 (1990 - 1991)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent
Total Phosphorus
SYSTEM # 1 (1992 - 1993)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent
Total Phosphorus
SYSTEM # 1 (1991 - 1992)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent
Total Phosphorus
SYSTEM # 1 (1993 - 1994)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

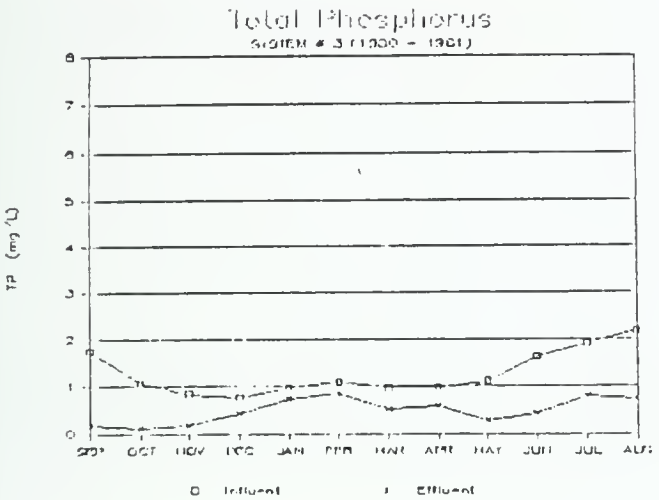
Figure A.11 Monthly average total phosphorus concentrations in Systems 1-5 (4 years).

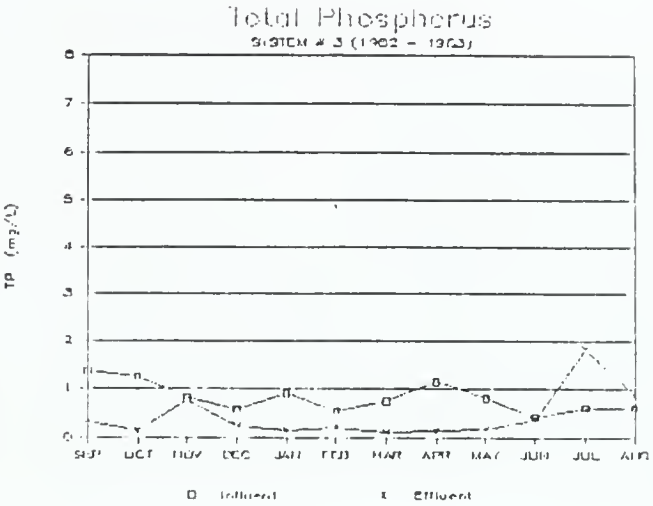

Total Phosphorus
SYSTEM # 2 (1900 - 1901)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

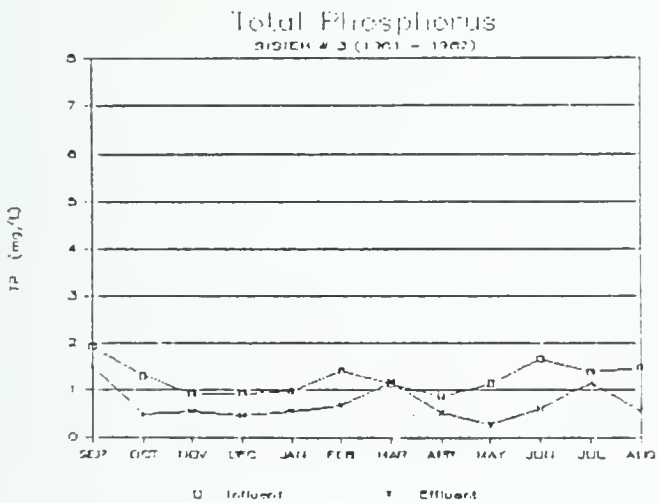

Total Phosphorus
SYSTEM # 2 (1902 - 1903)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
0 Influent
X Effluent

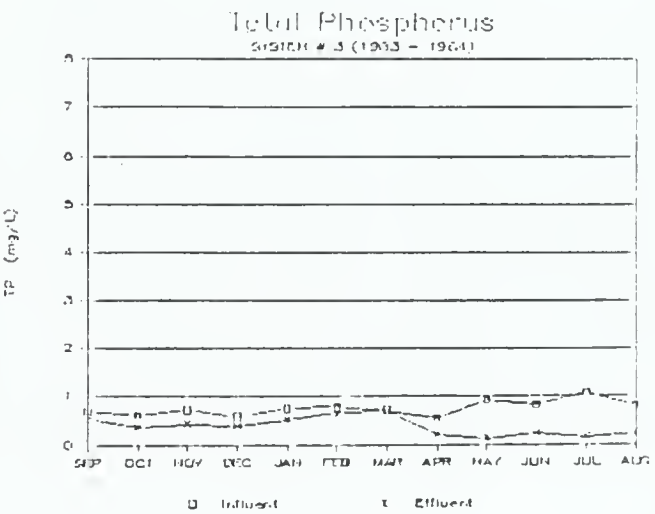

Total Phosphorus
SYSTEM # 2 (1901 - 1902)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
0 Influent
X Effluent

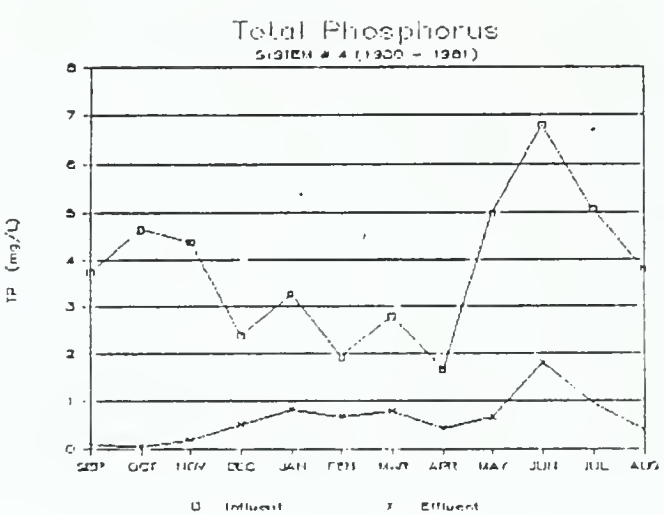

Total Phosphorus
SYSTEM # 2 (1903 - 1904)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
0 Influent
X Effluent

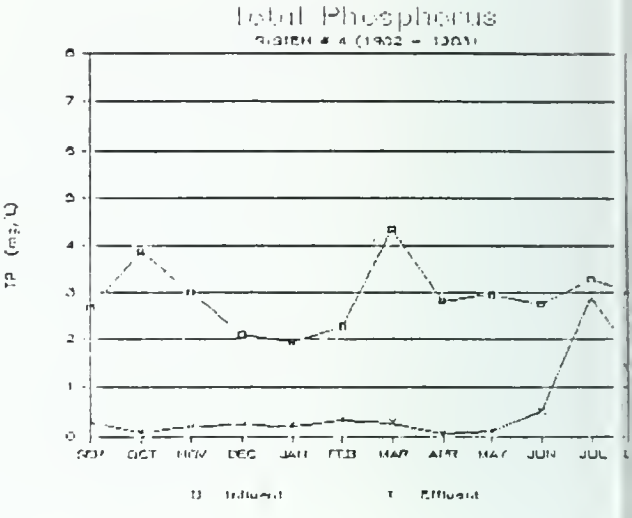

Total Phosphorus
SYSTEM # 3 (1980 - 1981)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

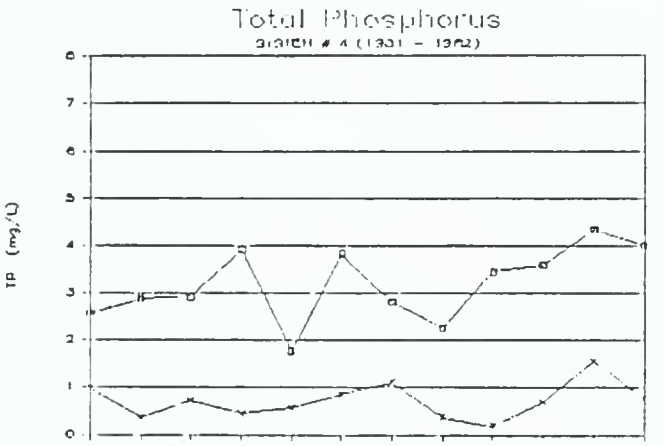

Total Phosphorus
SYSTEM # 3 (1982 - 1983)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

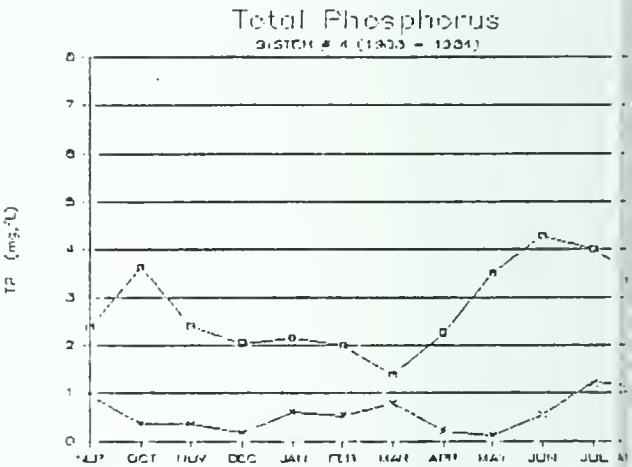

Total Phosphorus
SYSTEM # 3 (1981 - 1982)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

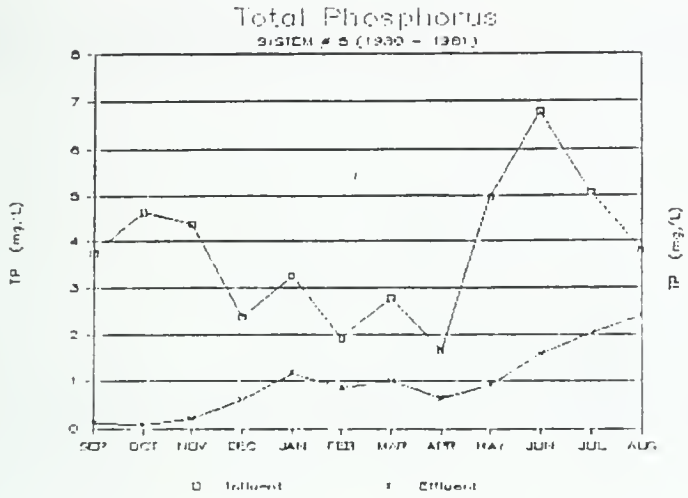

Total Phosphorus
SYSTEM # 3 (1983 - 1984)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

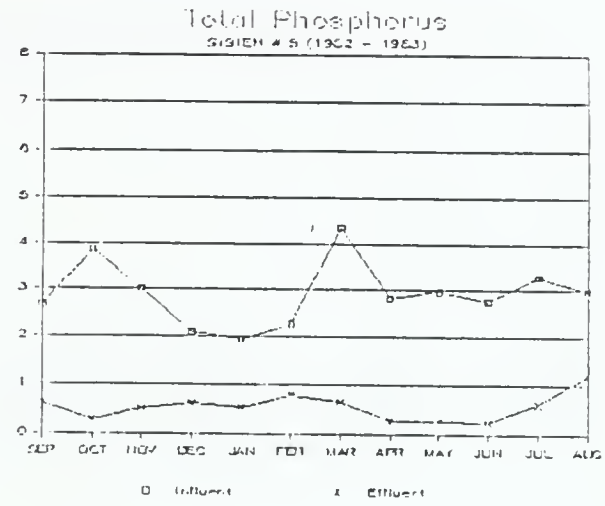

Total Phosphorus
SIGTEM # 4 (1980 - 1981)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

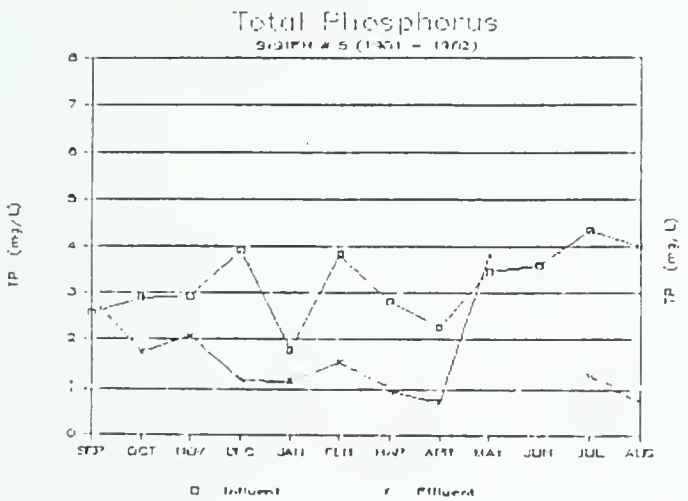

Total Phosphorus
SIGTEM # 4 (1982 - 1983)
TP (mg/L)
O Influent
X Effluent

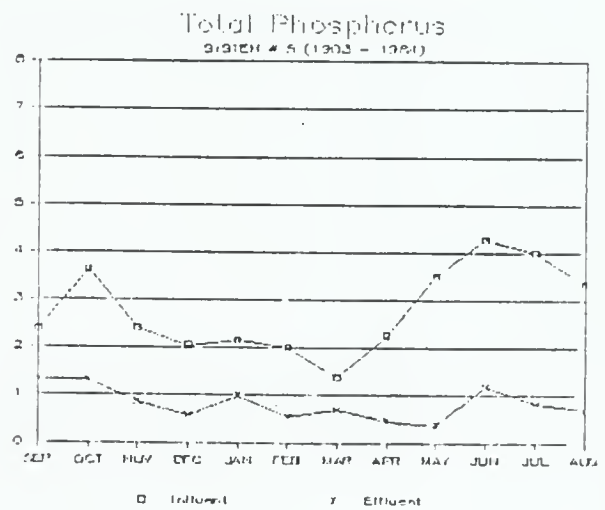

Total Phosphorus
SIGTEM # 4 (1981 - 1982)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Total Phosphorus
SIGTEM # 4 (1983 - 1984)
TP (mg/L)
O Influent
X Effluent


Total Phosphorus
SYSTEM # 5 (1980 - 1981)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Total Phosphorus
SYSTEM # 5 (1982 - 1983)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Total Phosphorus
SYSTEM # 5 (1981 - 1982)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Total Phosphorus
SYSTEM # 5 (1983 - 1984)
TP (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

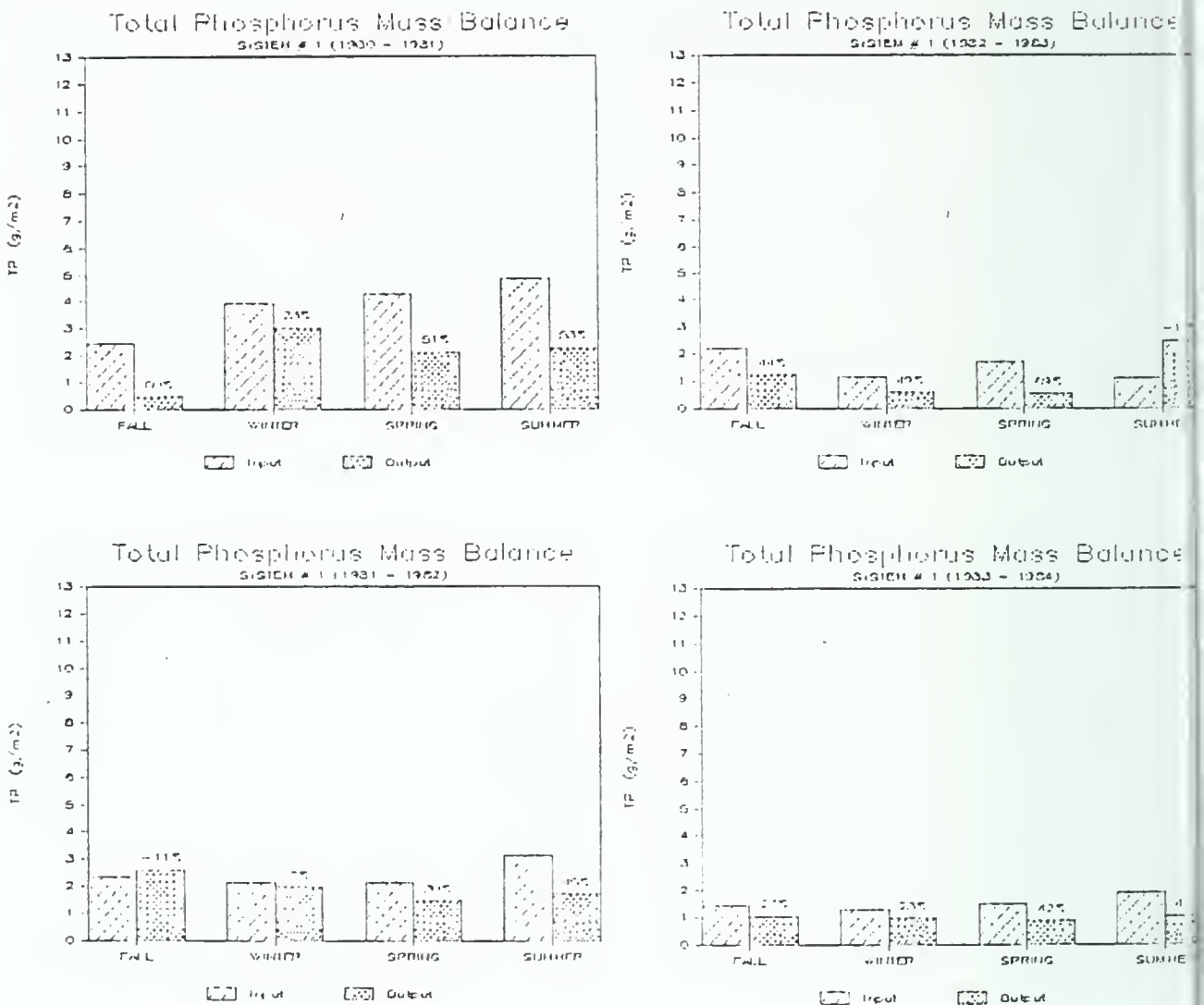
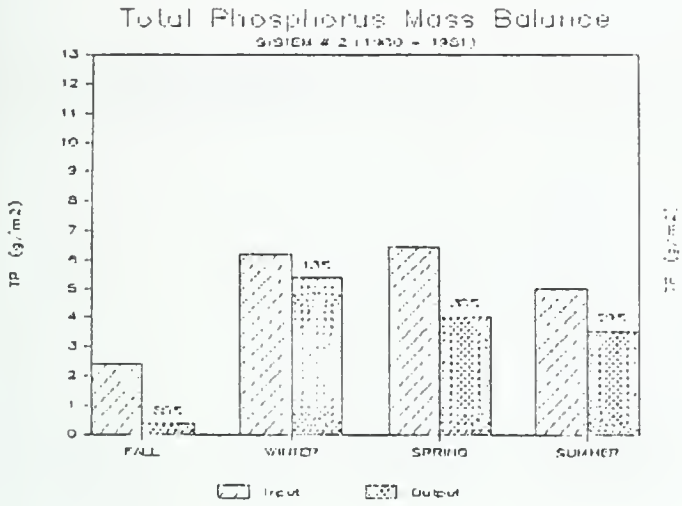
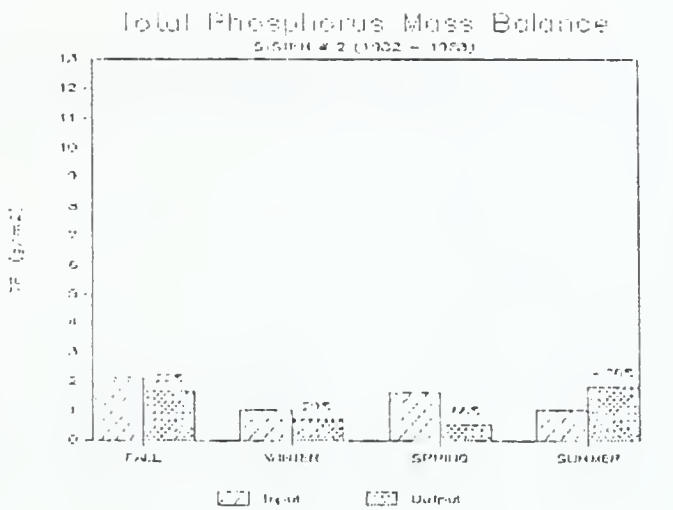
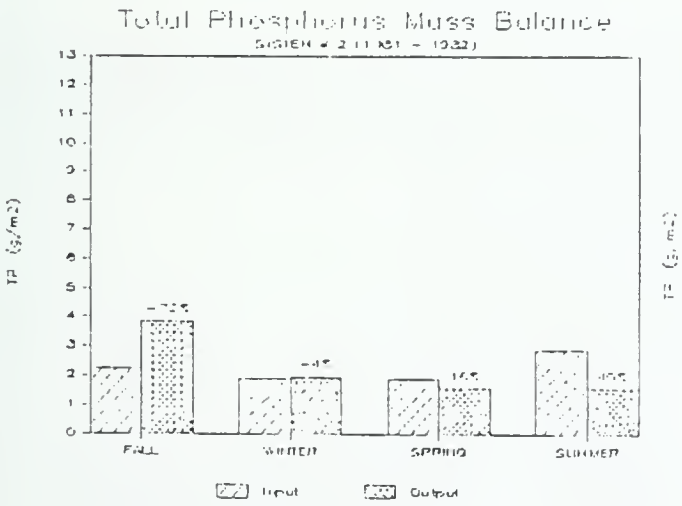
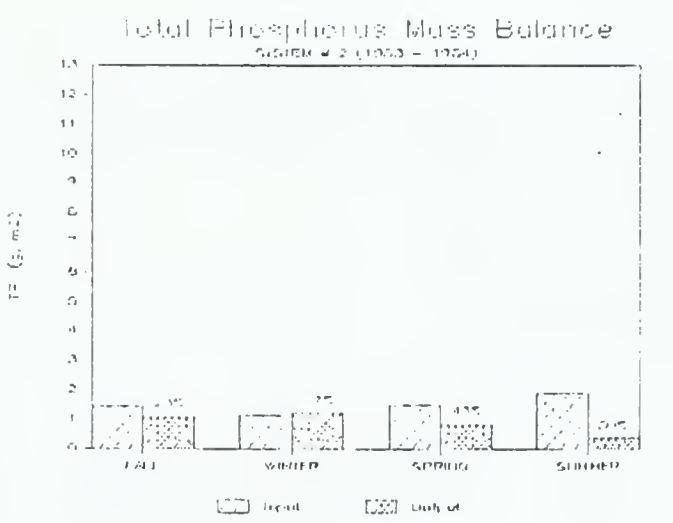


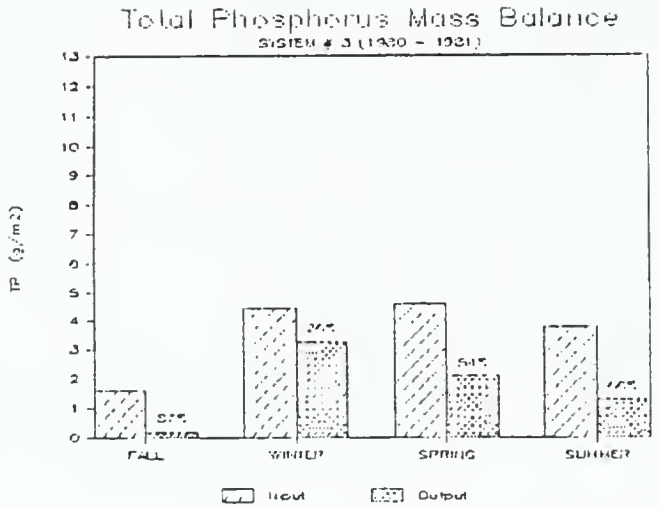
Figure A.12 Seasonal total phosphorus mass balances and removal efficiencies in Systems 1-5 (4 years).

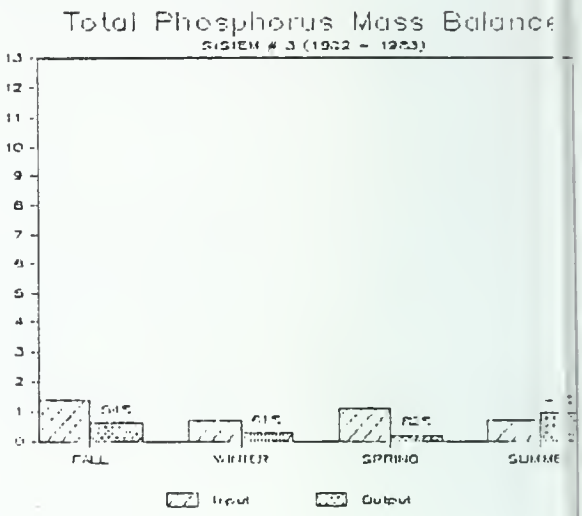

Total Phosphorus Mass Balance
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

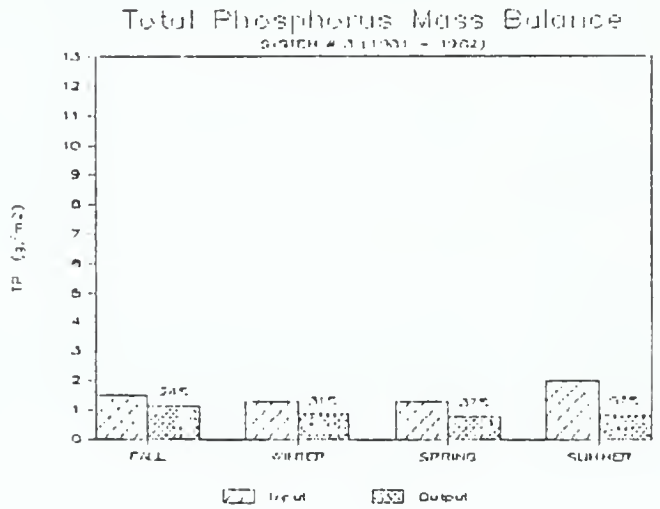

Total Phosphorus Mass Balance
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

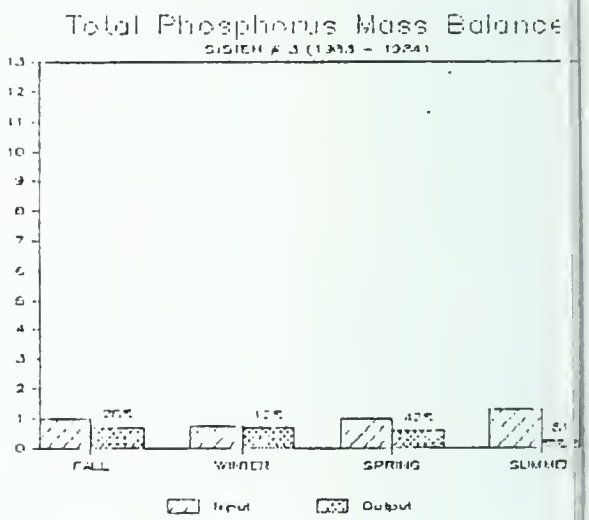

Total Phosphorus Mass Balance
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

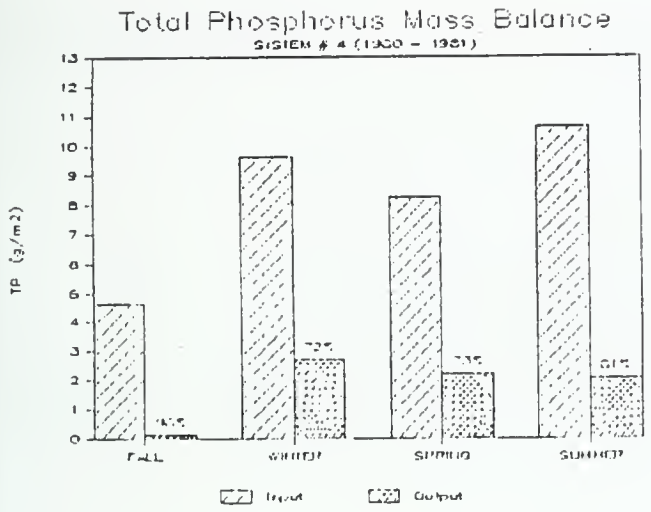

Total Phosphorus Mass Balance
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

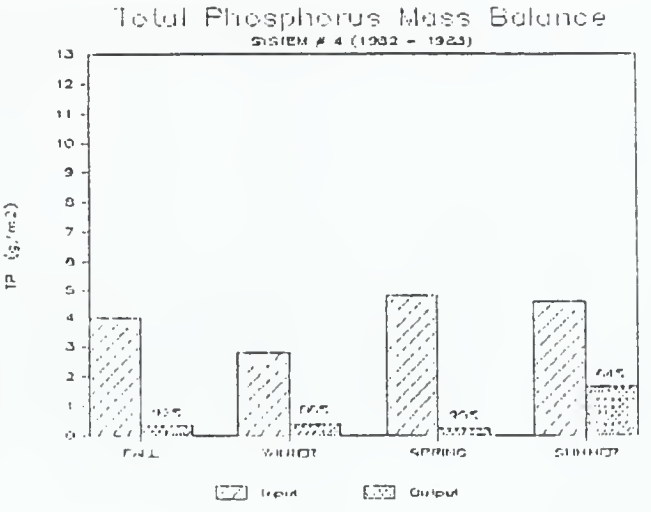

Total Phosphorus Mass Balance
SIGIEM # 3 (1980 - 1981)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

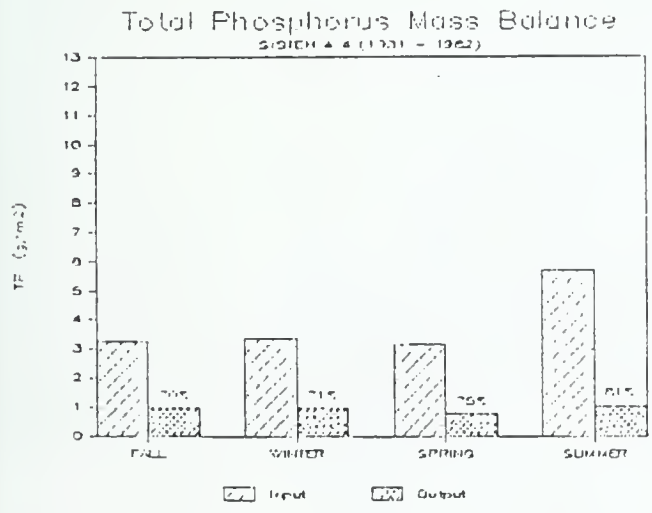

Total Phosphorus Mass Balance
SIGIEM # 3 (1982 - 1983)
TP (g/m2)
FALL
WINTER
SPRING
Input
Output

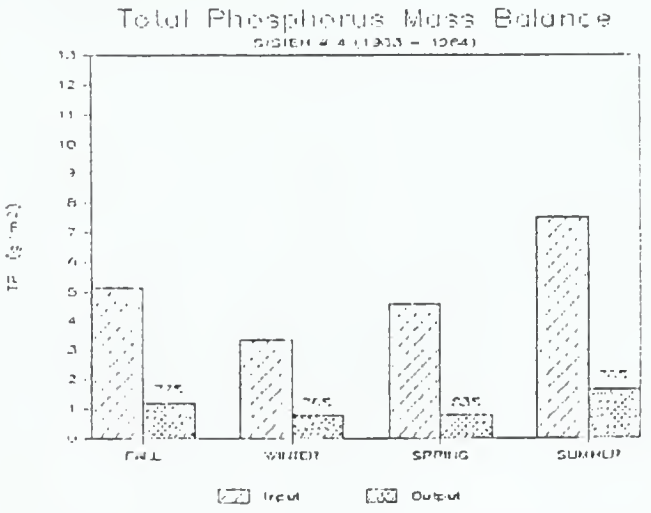

Total Phosphorus Mass Balance
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

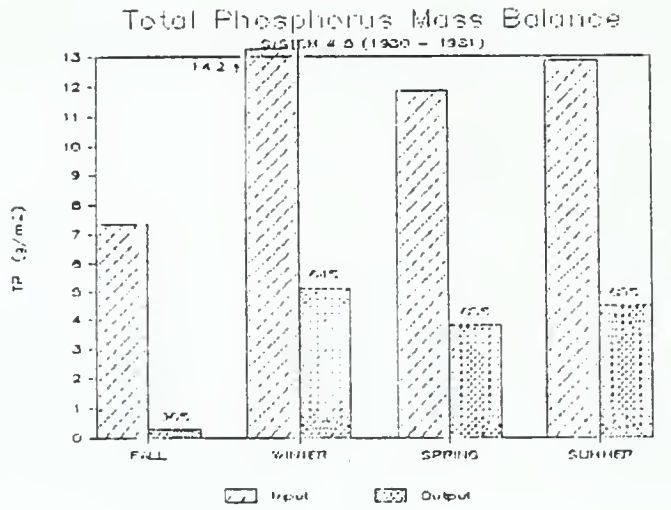

Total Phosphorus Mass Balance
SIGIEM # 3 (1983 - 1984)
TP (g/m2)
FALL
WINTER
SPRING
Input
Output

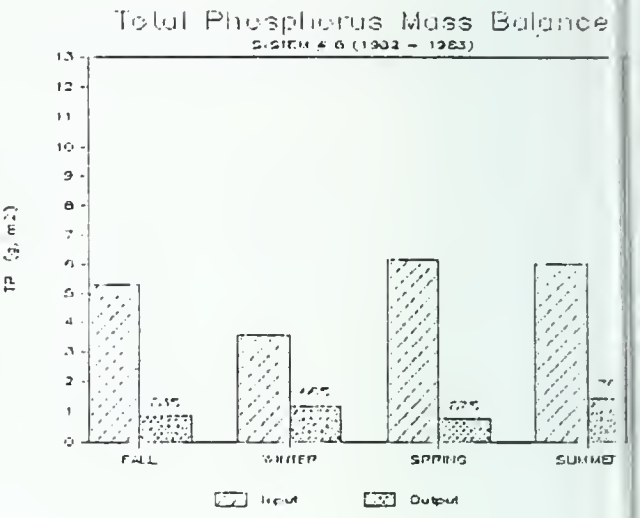

Total Phosphorus Mass Balance
SYSTEM # 4 (1980 - 1981)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

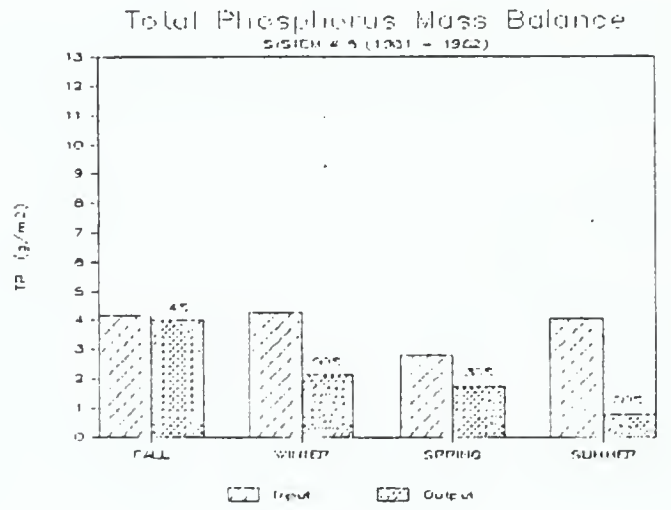

Total Phosphorus Mass Balance
SYSTEM # 4 (1982 - 1983)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

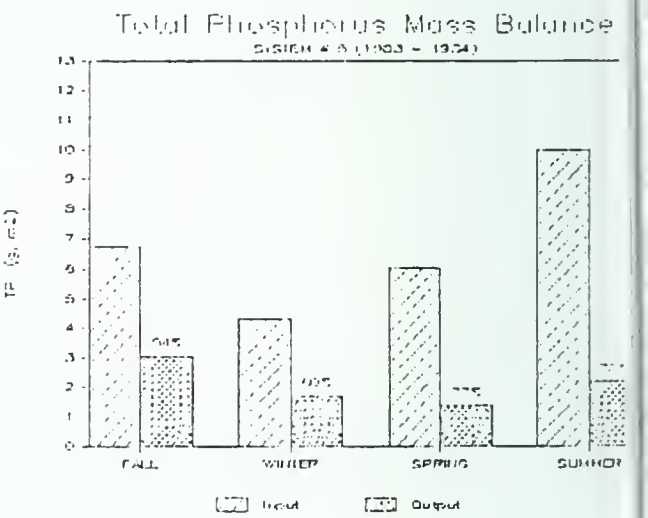

Total Phosphorus Mass Balance
SYSTEM # 4 (1981 - 1982)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Phosphorus Mass Balance
SYSTEM # 4 (1983 - 1984)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Phosphorus Mass Balance
SISTER # 6 (1930 - 1931)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Phosphorus Mass Balance
SISTER # 6 (1932 - 1933)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Phosphorus Mass Balance
SISTER # 6 (1933 - 1934)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Phosphorus Mass Balance
SISTER # 6 (1933 - 1934)
TP (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

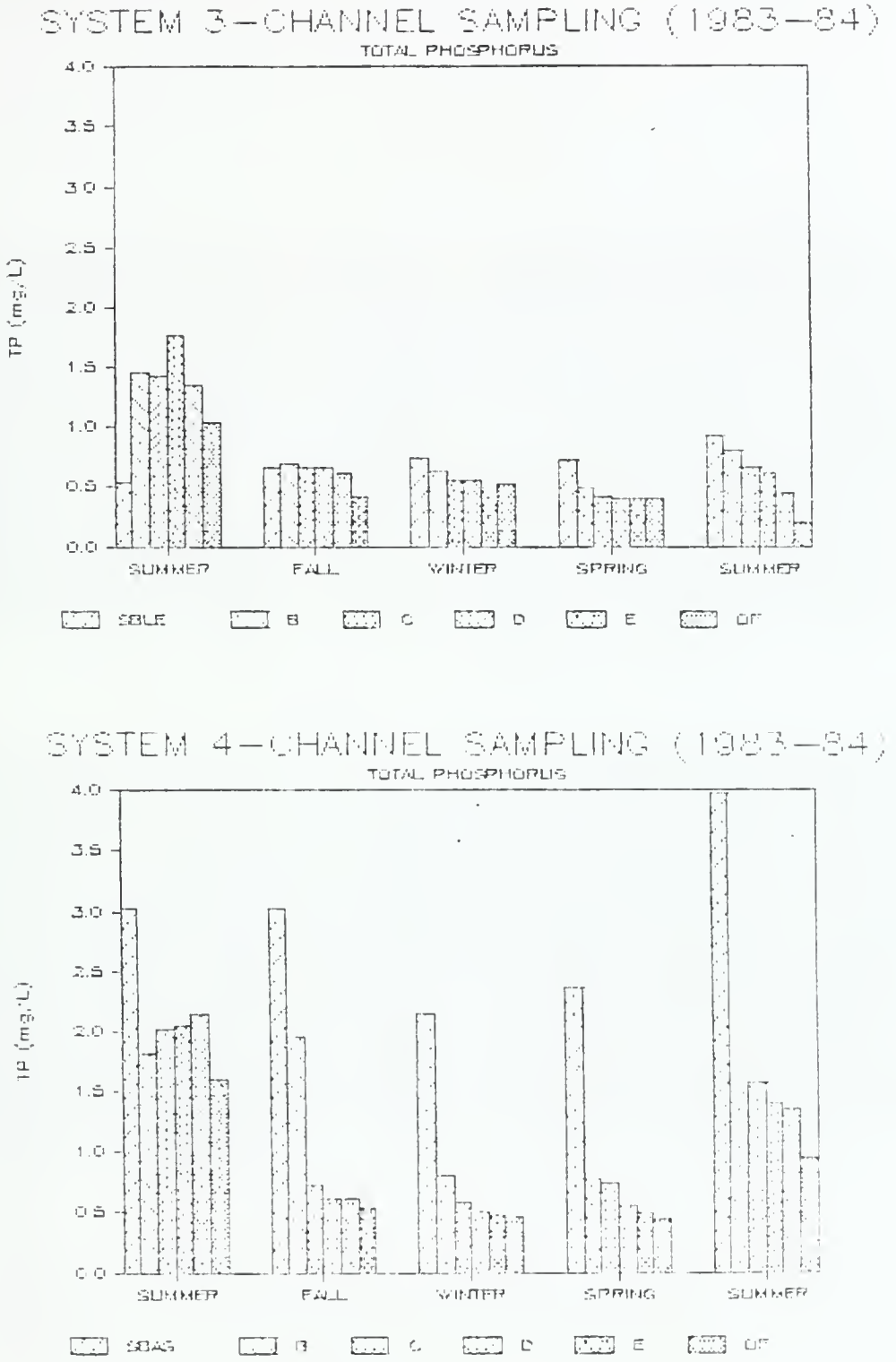


Figure A.13 Seasonal total phosphorus concentrations in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of Systems 3 and 4 (1983-4).

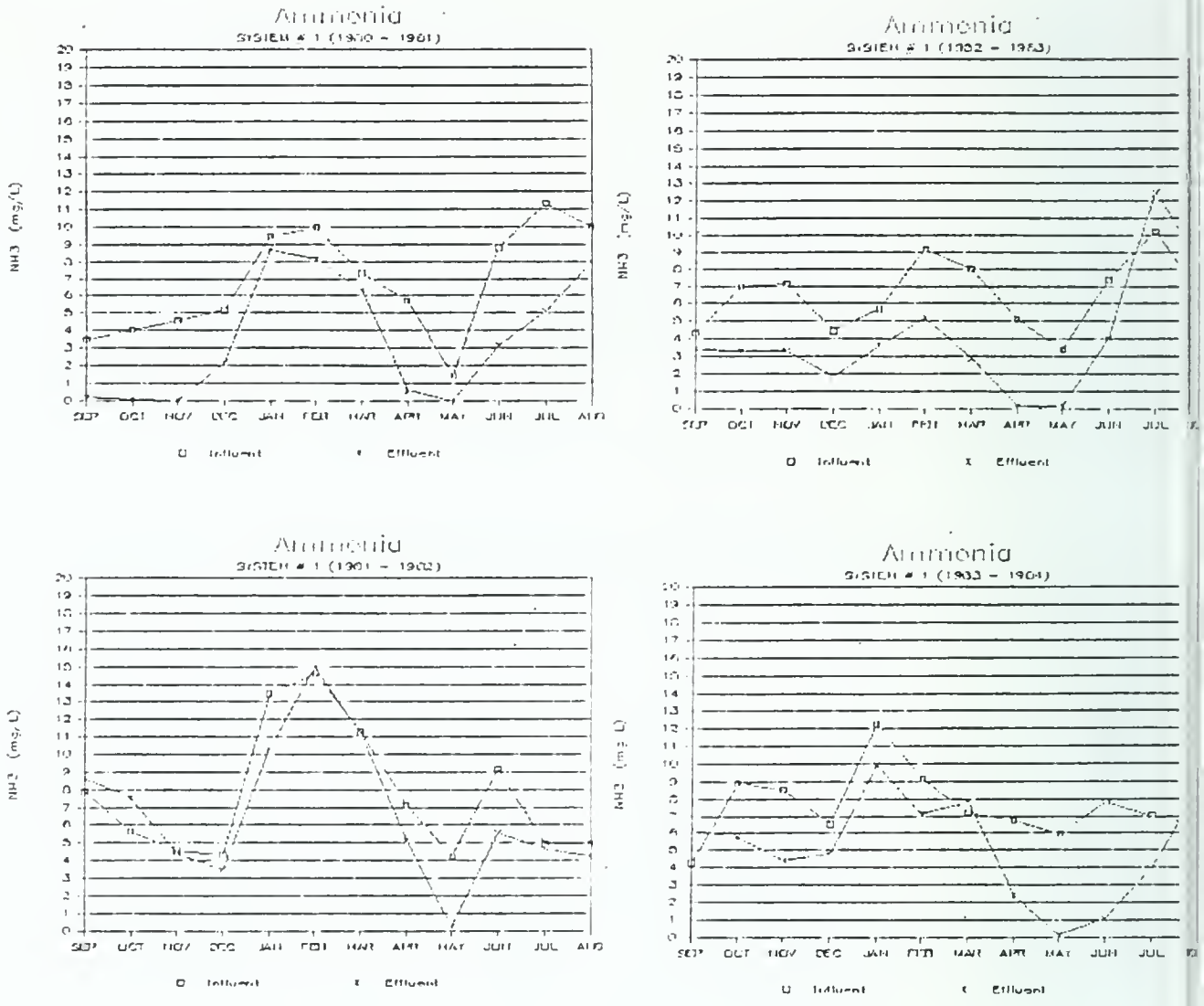
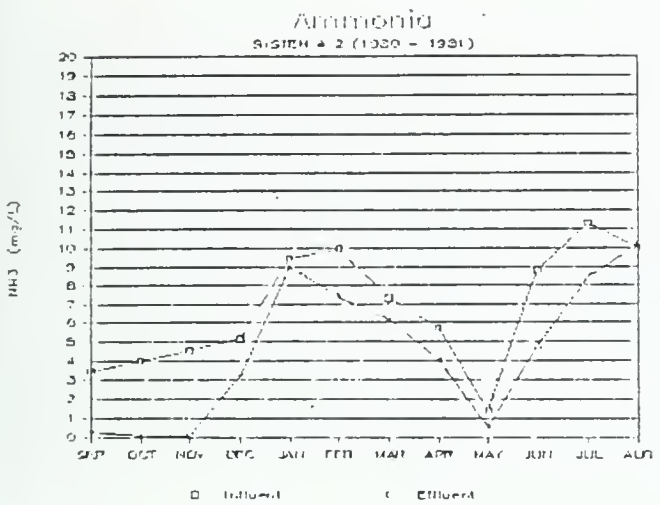
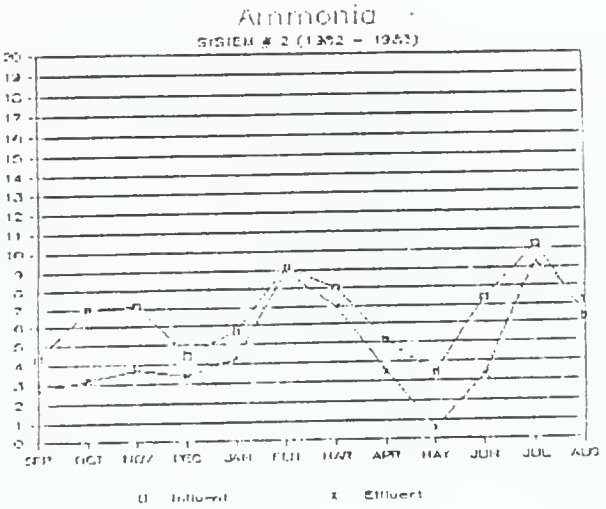
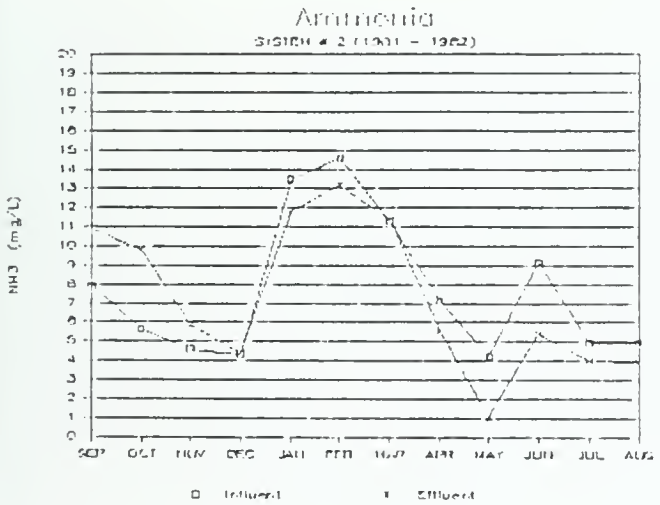
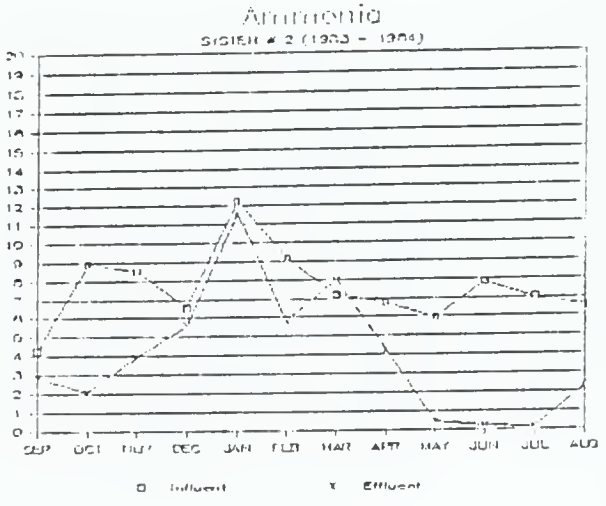


Figure A.14 Monthly average ammonia concentrations in Systems 1-5 (4 years).

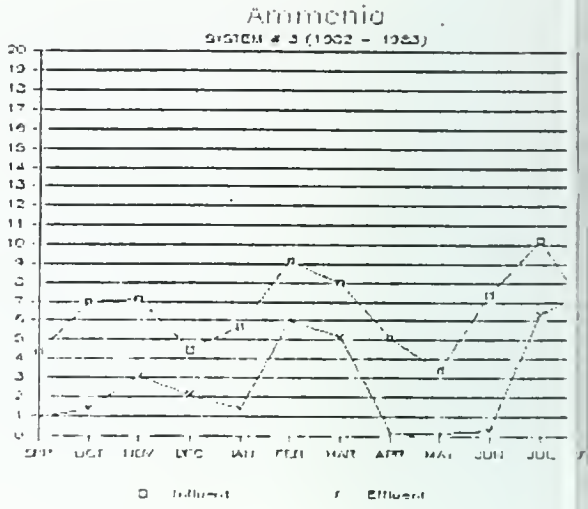

Ammonia
SISIEM # 2 (1990 - 1991)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

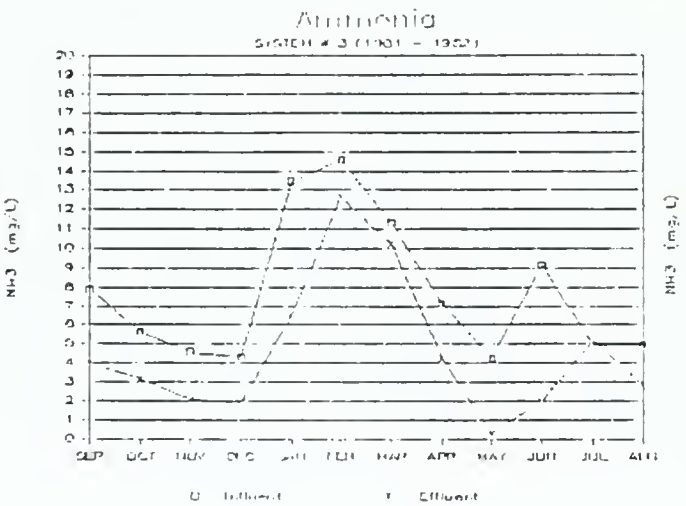

Ammonia
SISIEM # 2 (1992 - 1993)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

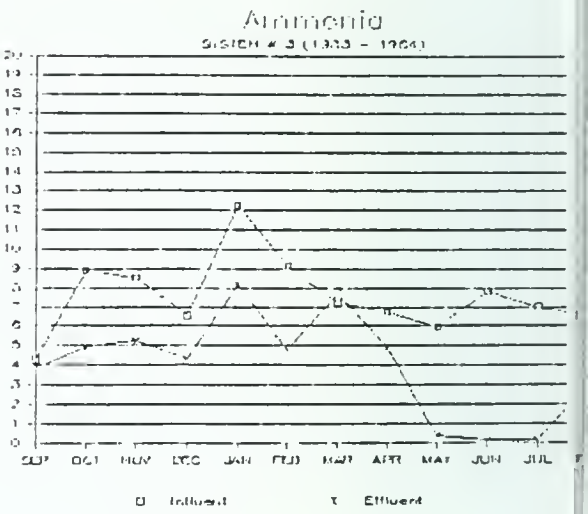

Ammonia
SISIEM # 2 (1991 - 1992)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

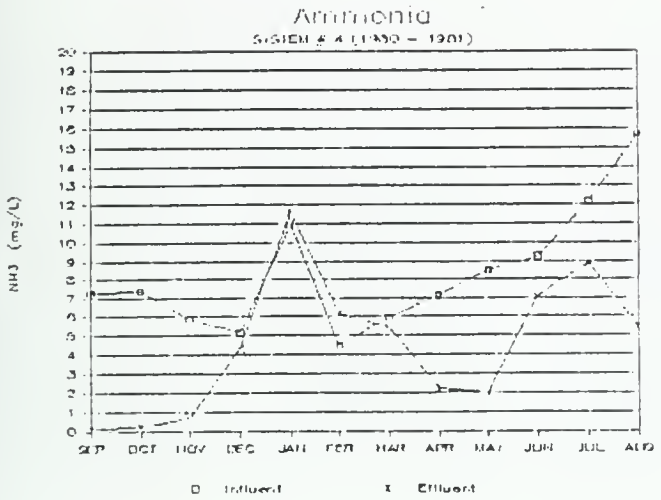

Ammonia
SISIEM # 2 (1993 - 1994)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

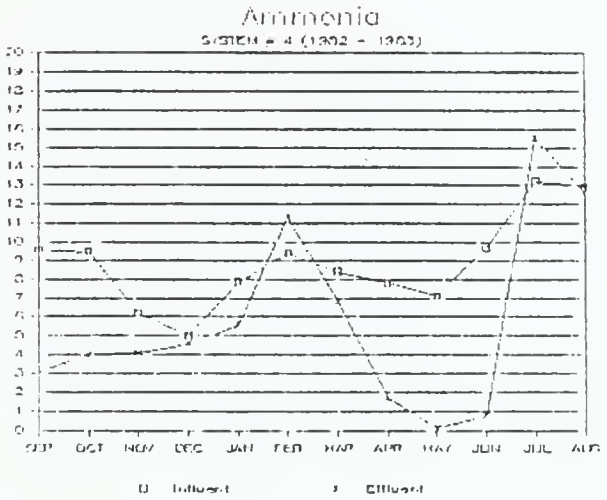
Ammonia
SISTEM # 3 (1980 - 1981)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
D Influent
E Effluent

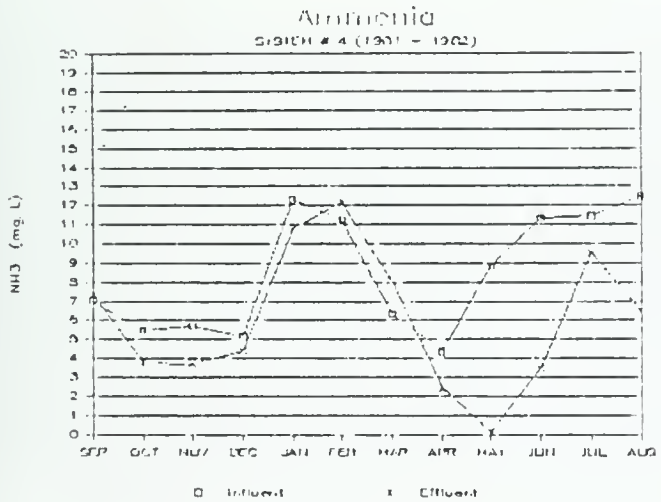

Ammonia
SISTEM # 3 (1982 - 1983)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
D Influent
E Effluent

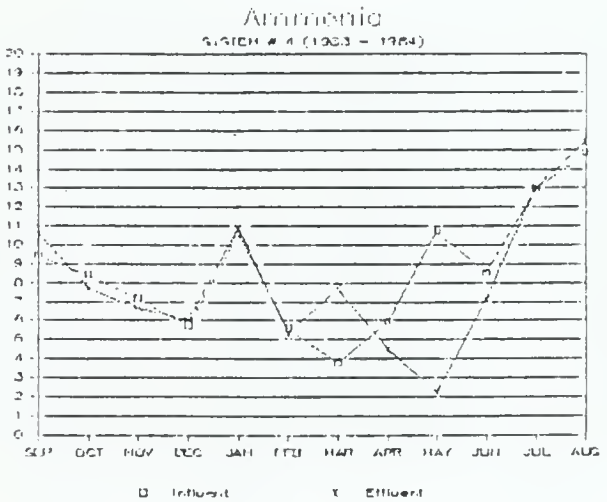

Ammonia
SISTEM # 3 (1981 - 1982)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
D Influent
E Effluent

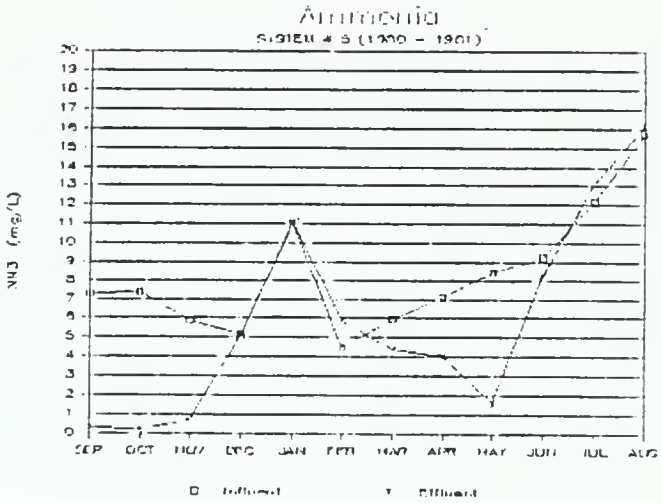

Ammonia
SISTEM # 3 (1983 - 1984)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
D Influent
E Effluent

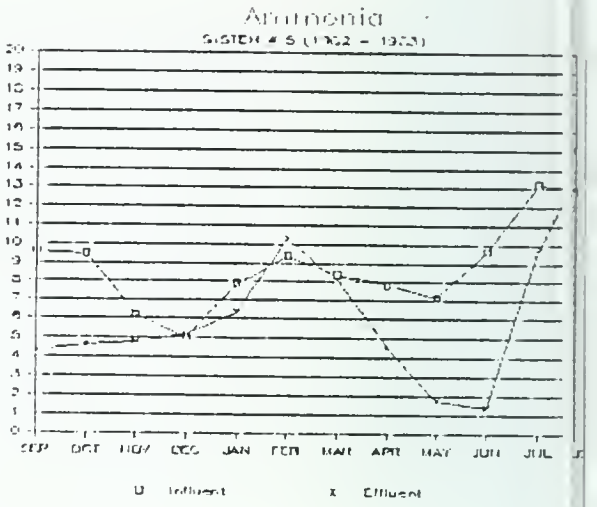

Ammonia
SYSTEM # 4 (1900 - 1901)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

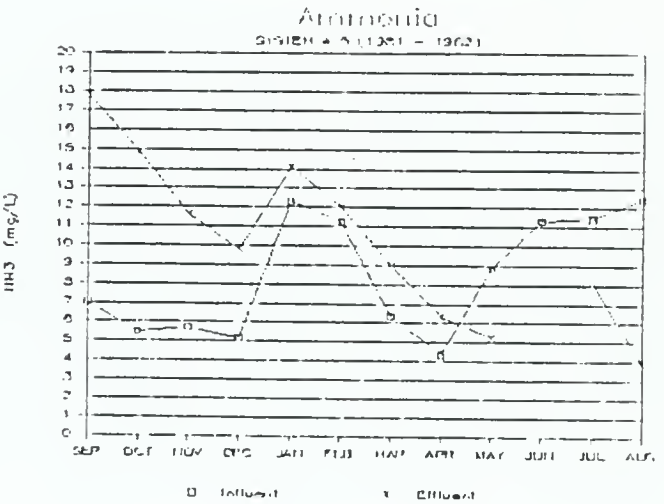

Ammonia
SYSTEM # 4 (1902 - 1903)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

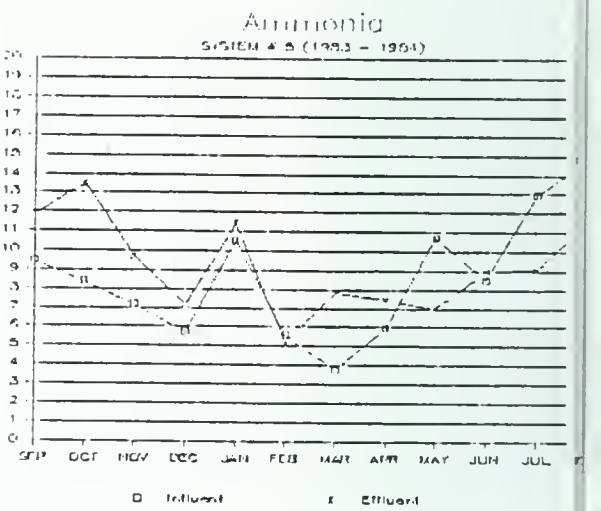

Ammonia
SYSTEM # 4 (1901 - 1902)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Ammonia
SYSTEM # 4 (1903 - 1904)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Ammonia
SYSTEM # 5 (1990 - 1991)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
D Influent
X Effluent


Ammonia
SYSTEM # 5 (1992 - 1993)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
D Influent
X Effluent


Ammonia
SYSTEM # 5 (1991 - 1992)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
D Influent
X Effluent


Ammonia
SYSTEM # 5 (1993 - 1994)
NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
D Influent
X Effluent

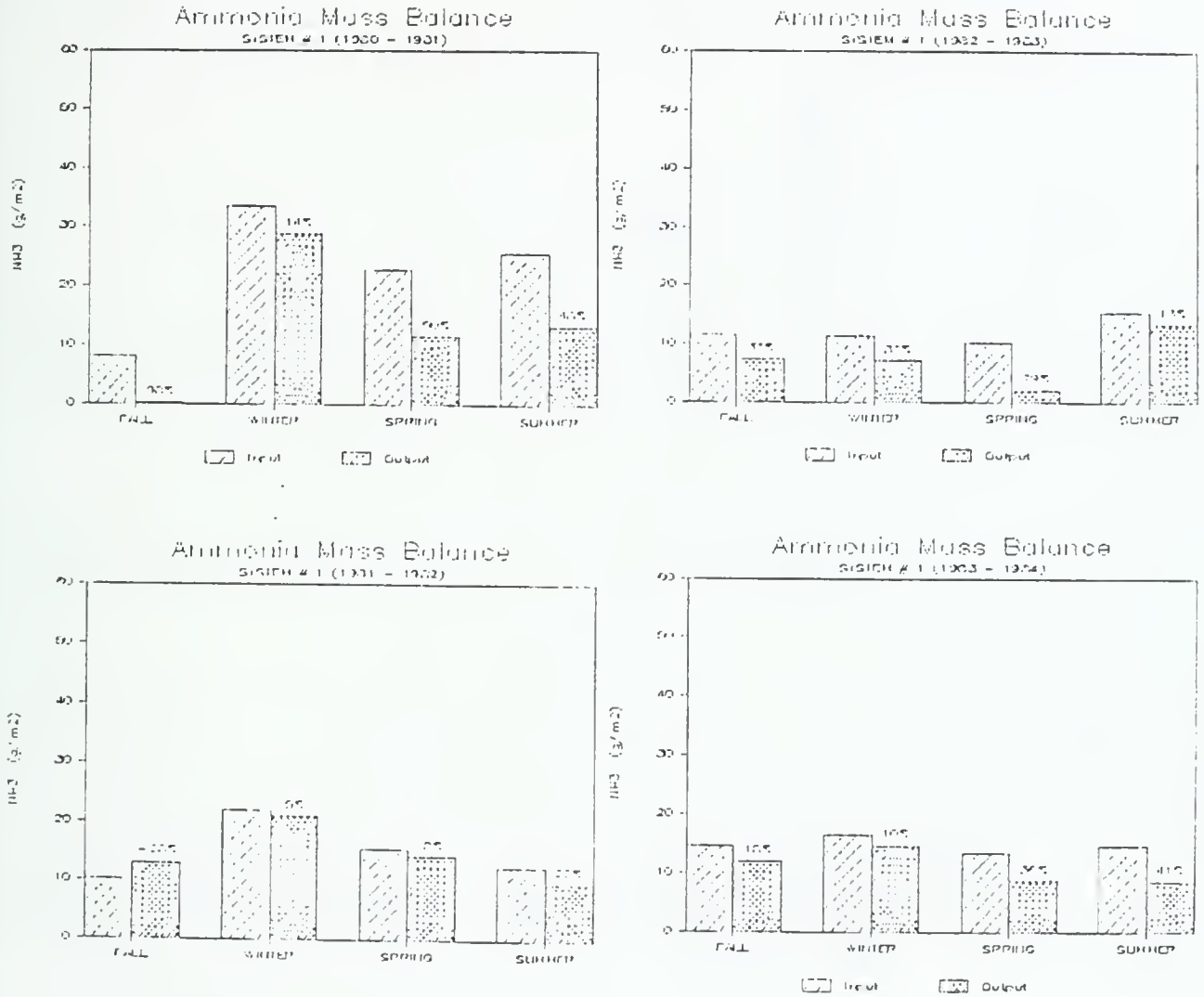
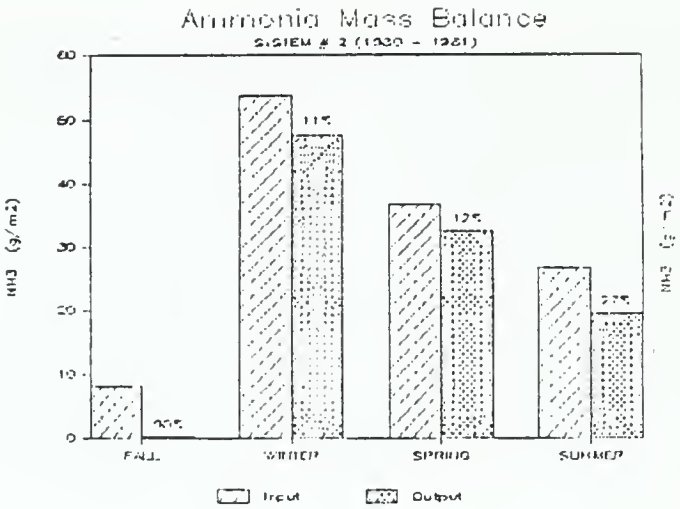
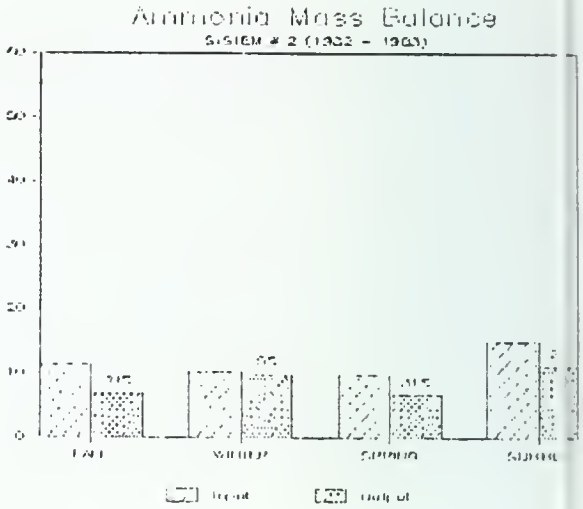
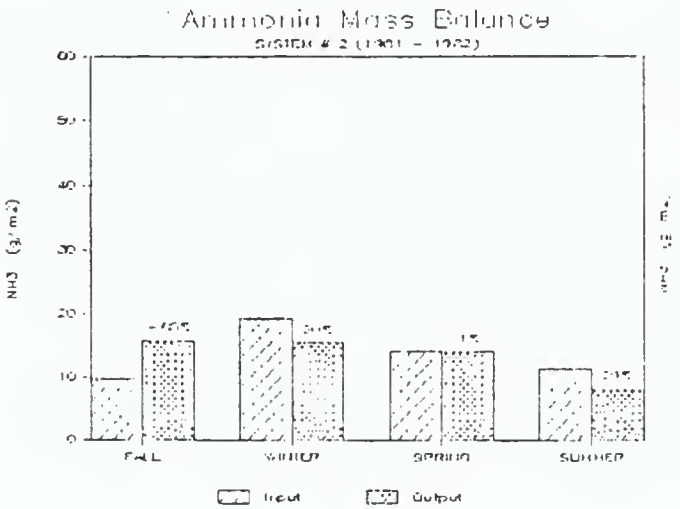
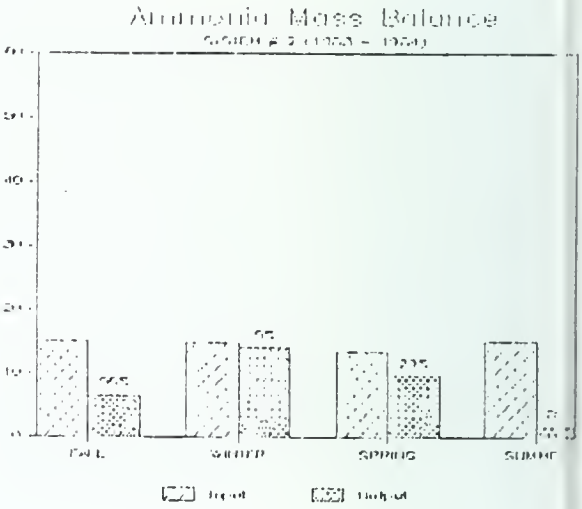


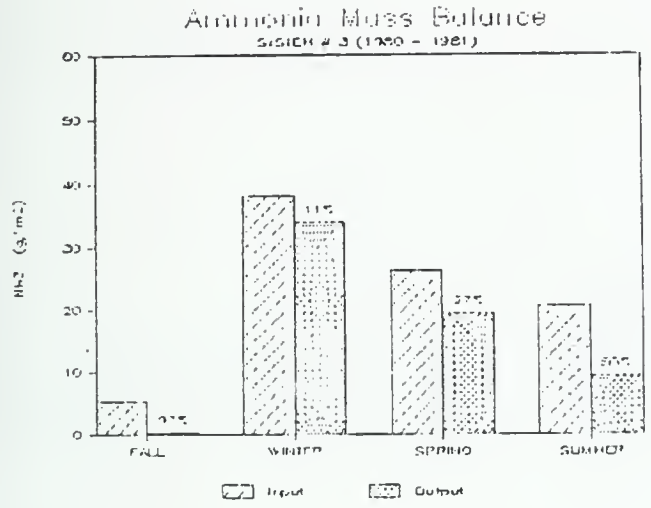
Figure A.15 Seasonal ammonia mass balances and removal efficiencies in Systems 1-5 (4 years).

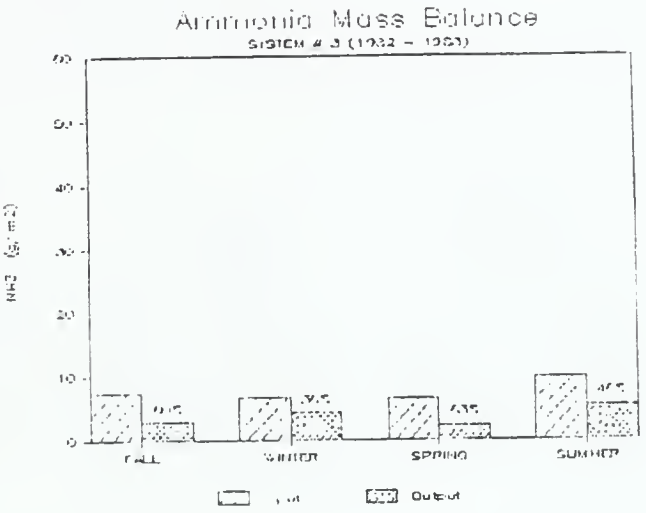

Ammonia Mass Balance
SYSTEM # 2 (1980 - 1981)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

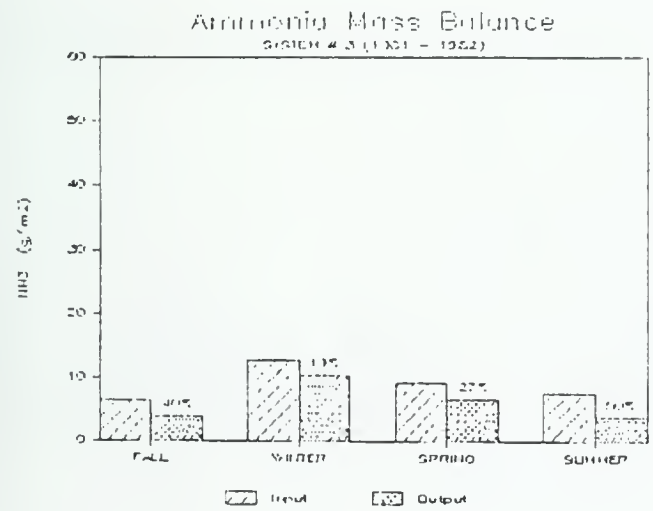

Ammonia Mass Balance
SYSTEM # 2 (1982 - 1983)

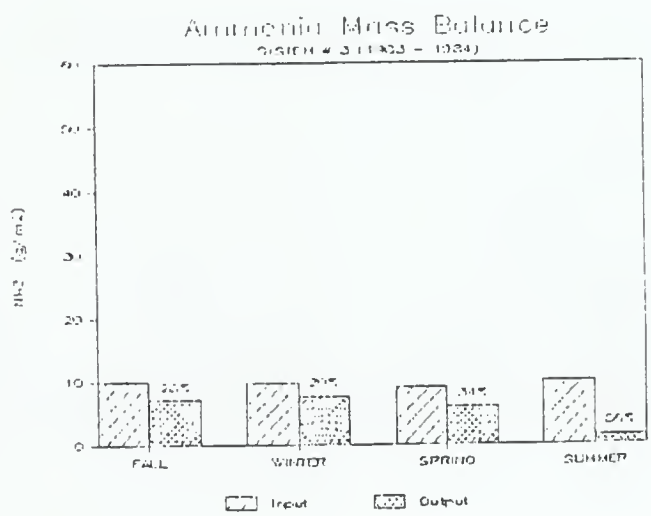

Ammonia Mass Balance
SYSTEM # 2 (1981 - 1982)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

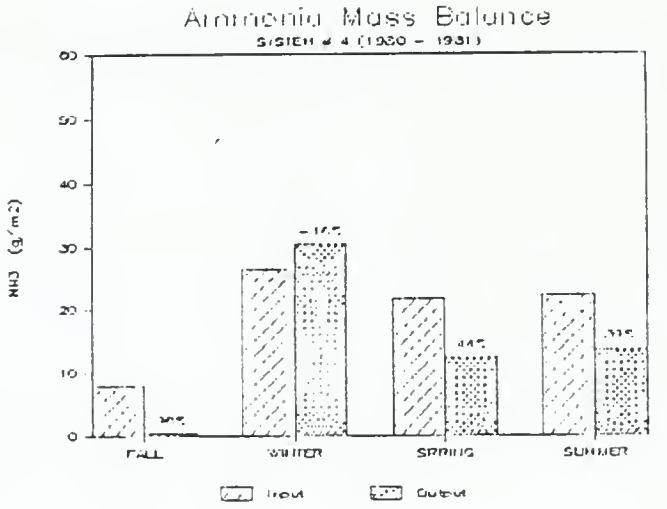

Ammonia Mass Balance

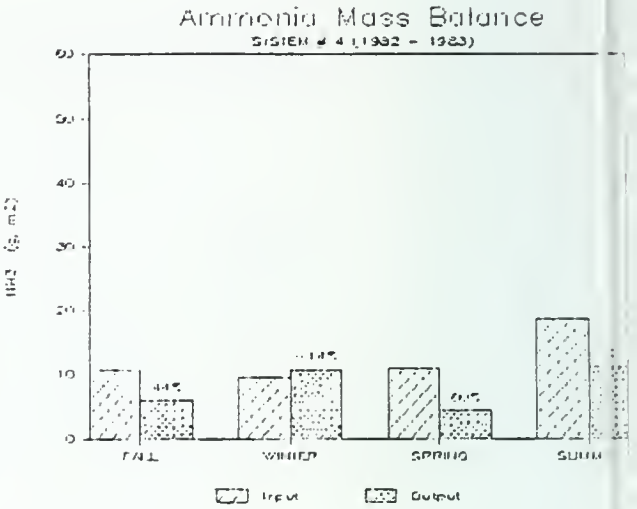

Ammonia Mass Balance
SISIEN # 3 (1980 - 1981)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

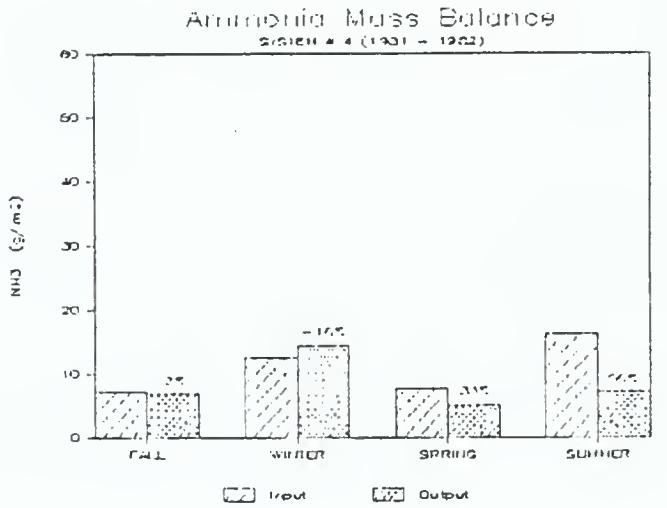

Ammonia Mass Balance
SISIEN # 3 (1982 - 1983)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Output

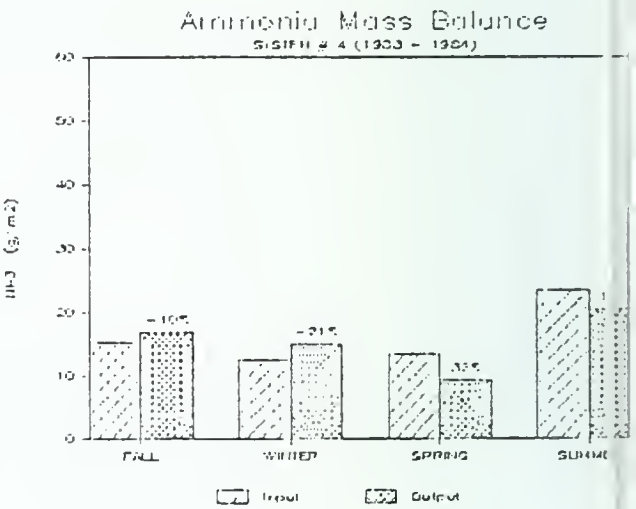

Ammonia Mass Balance
SISIEN # 3 (1981 - 1982)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

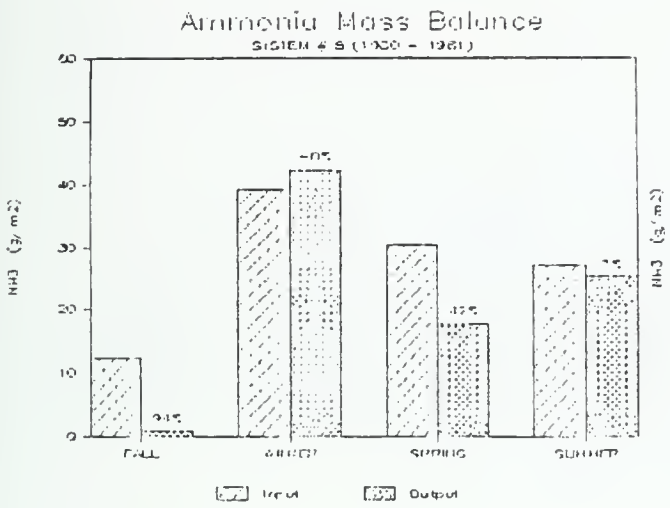

Ammonia Mass Balance
SISIEN # 3 (1983 - 1984)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

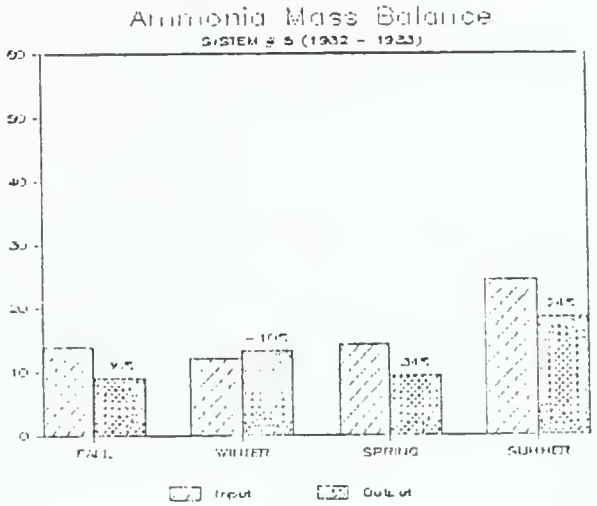

Ammonia Mass Balance
SISIEN # 4 (1980 - 1981)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

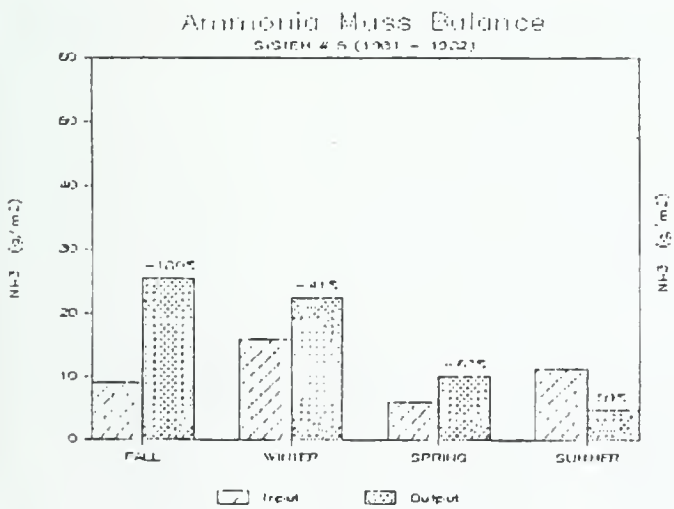

Ammonia Mass Balance
SISIEN # 4 (1982 - 1983)
NH3 (g/m2)
FALL
WINTER
SPRING
Input
Output

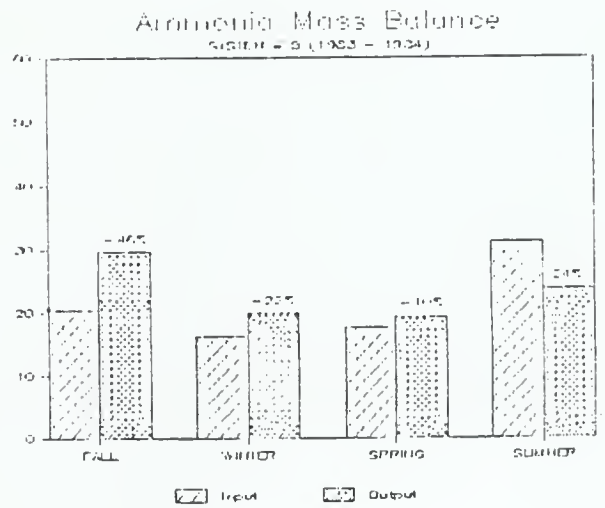

Ammonia Mass Balance
SISIEN # 4 (1981 - 1982)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

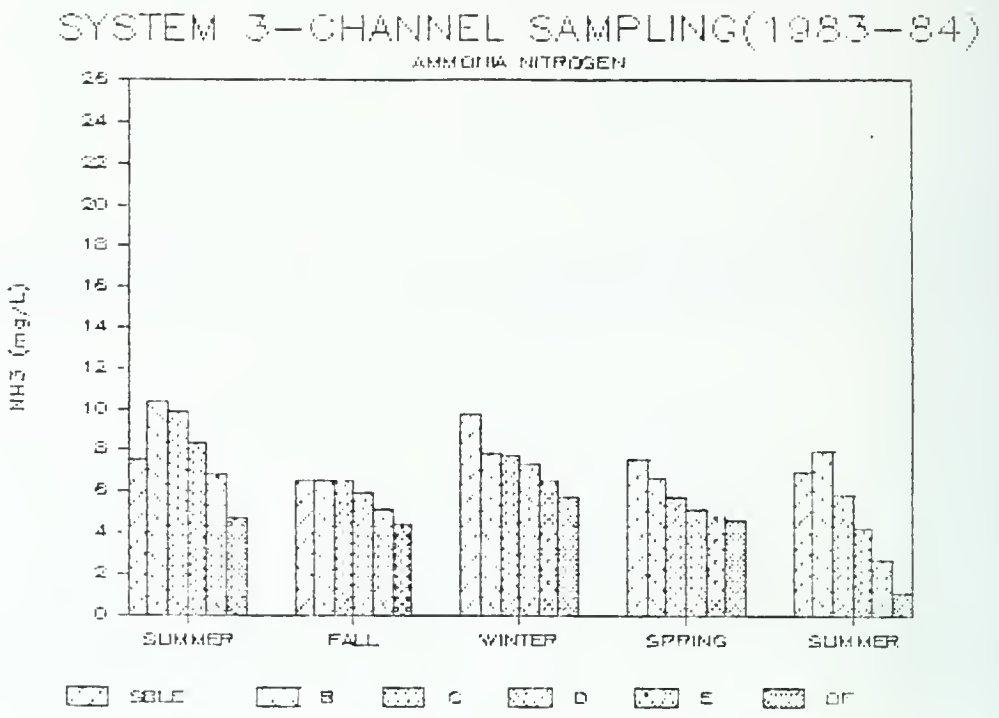

Ammonia Mass Balance
SISIEN # 4 (1983 - 1984)
NH3 (g/m2)
FALL
WINTER
SPRING
Input
Output


Ammonia Mass Balance
SYSTEM # 5 (1980 - 1981)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Ammonia Mass Balance
SYSTEM # 5 (1982 - 1983)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Ammonia Mass Balance
SYSTEM # 5 (1981 - 1982)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Ammonia Mass Balance
SYSTEM # 5 (1983 - 1984)
NH3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output



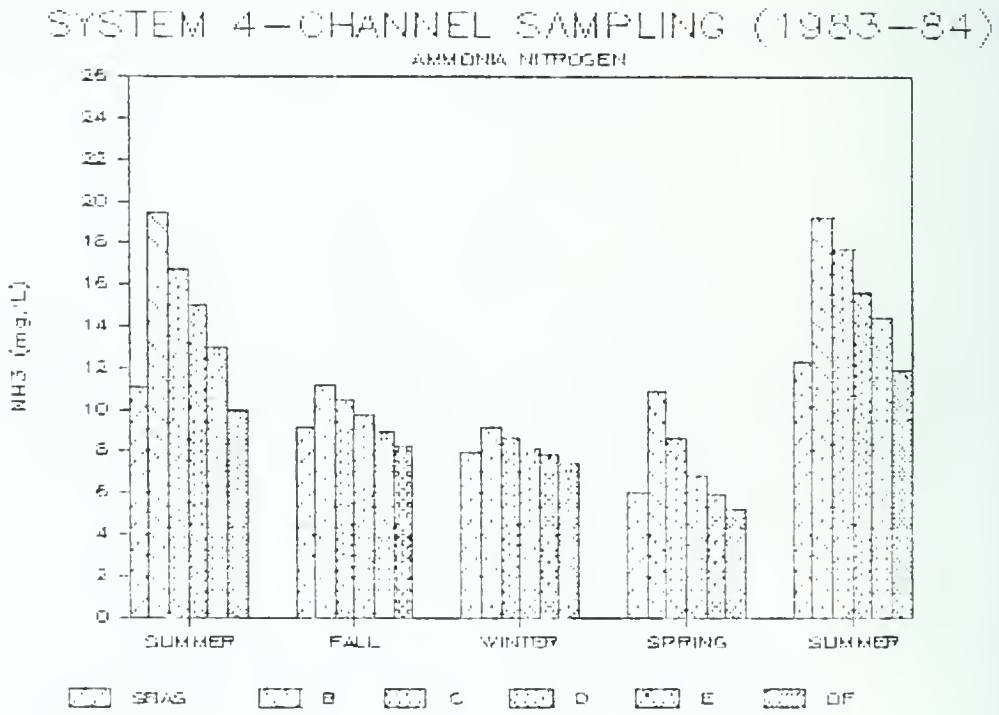


Figure A.16 Seasonal ammonia concentrations in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of System 3 and 4 (1983-4).

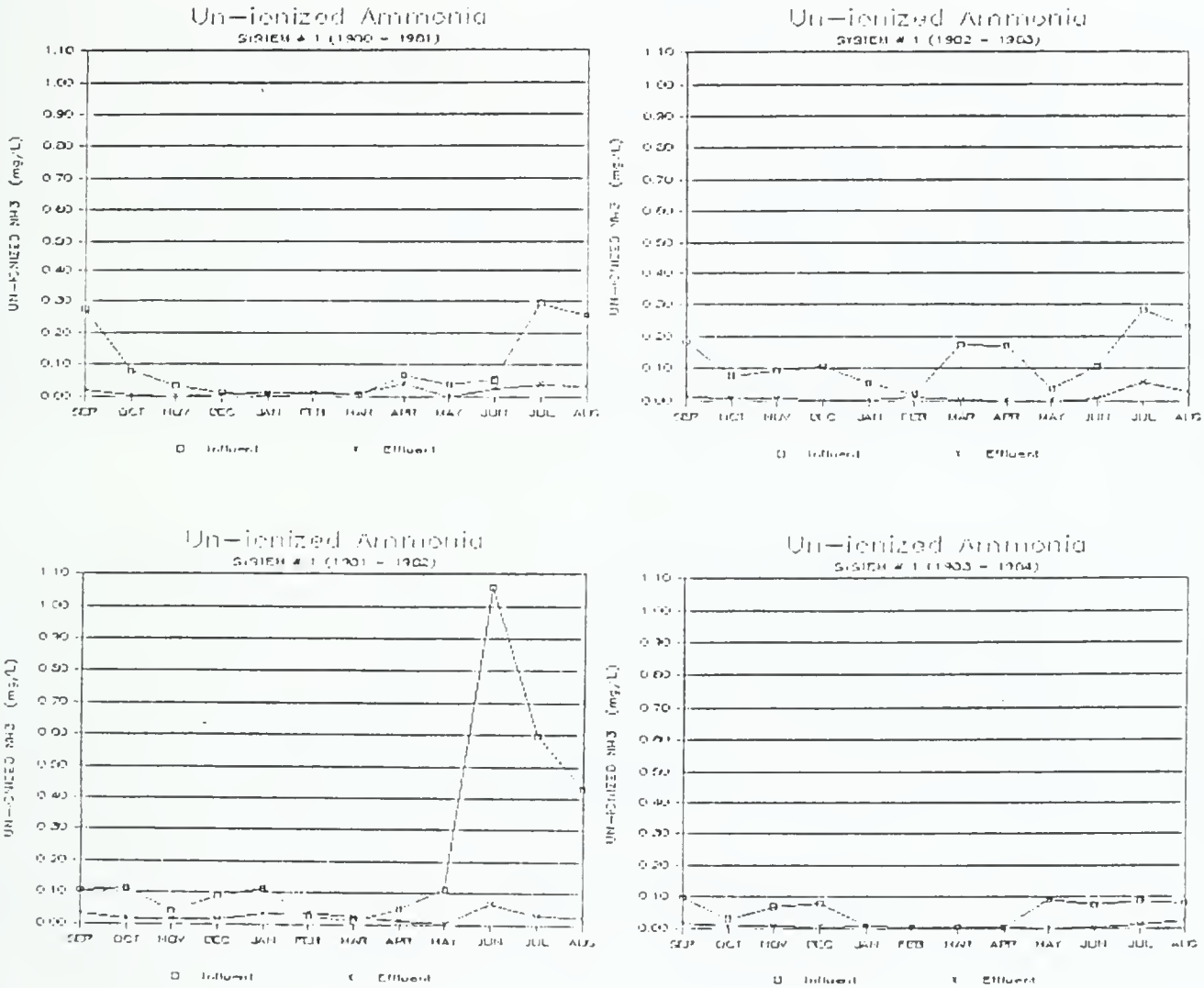
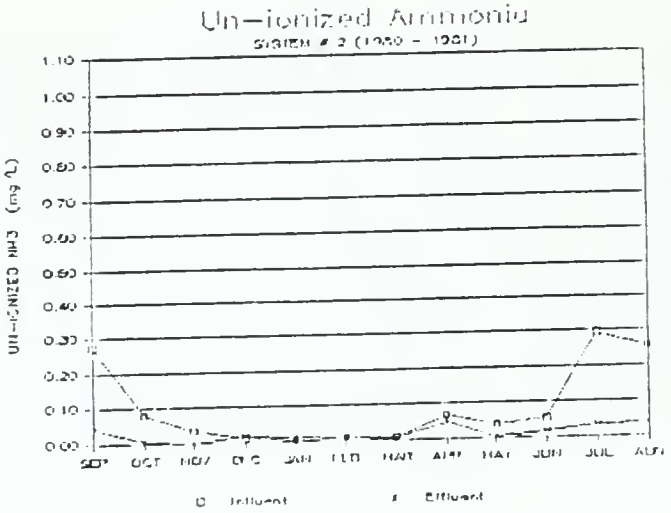
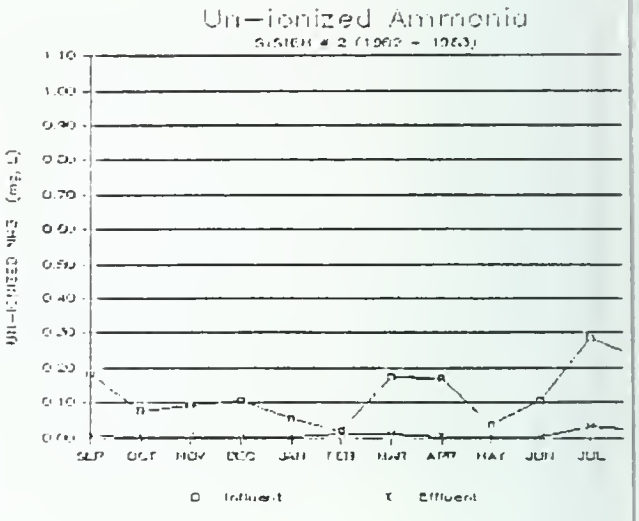
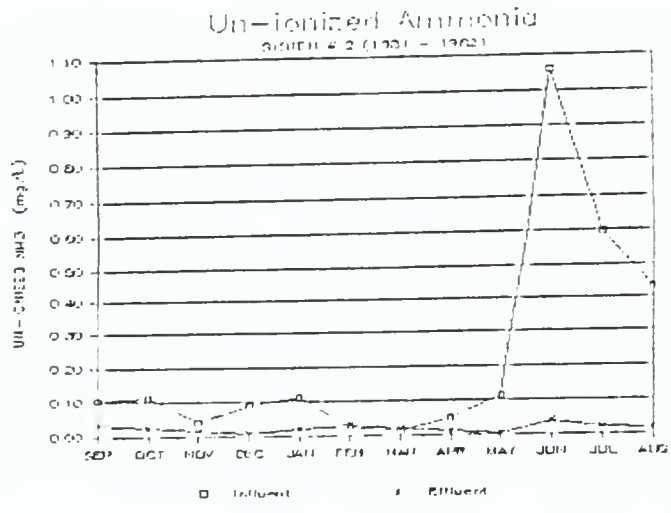
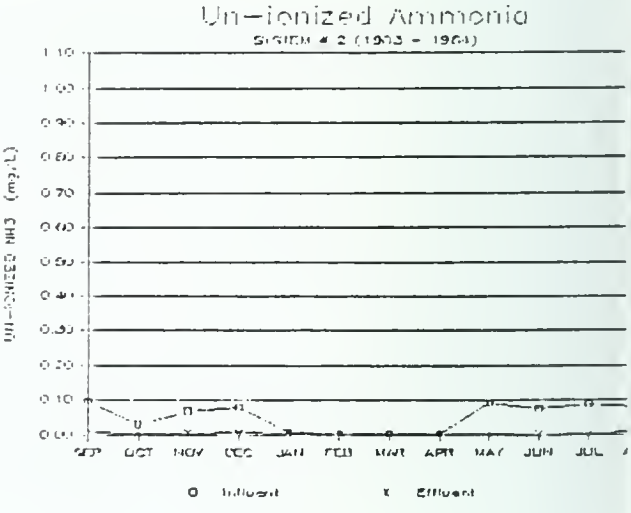

Un-ionized Ammonia
SYSTEM # 1 (1900 - 1901)
Un-ionized Ammonia
SYSTEM # 1 (1902 - 1903)
Un-ionized Ammonia
SYSTEM # 1 (1901 - 1902)
Un-ionized Ammonia
SYSTEM # 1 (1903 - 1904)
UN-IONIZED NH3 (mg/L)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

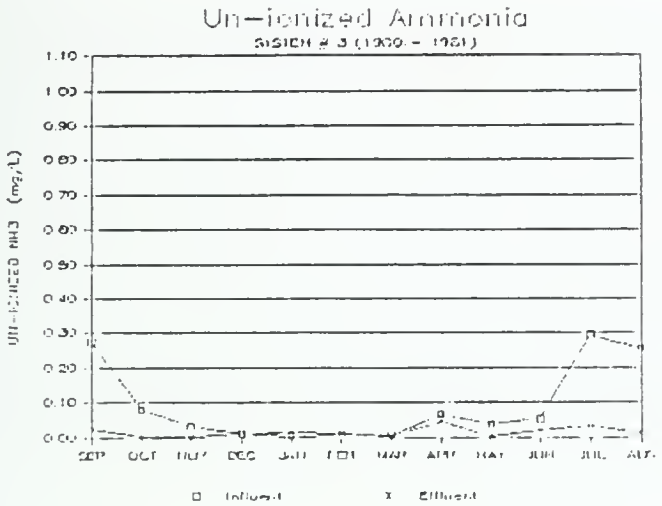
Figure A.17 Monthly average un-ionized ammonia concentrations in Systems 1-5 (4 years).

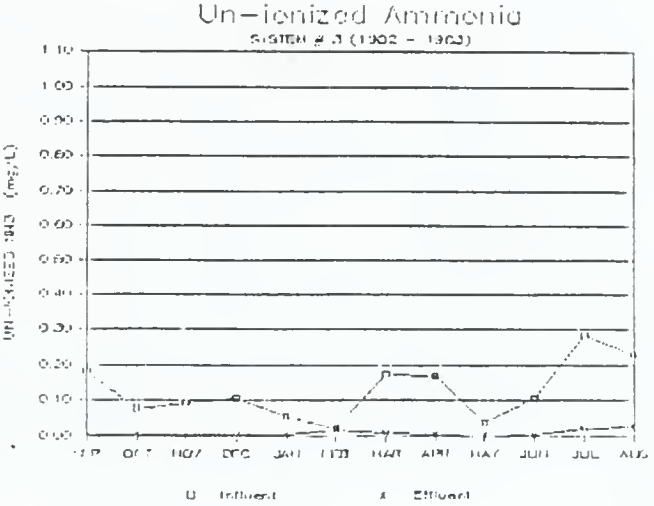

Un-ionized Ammonia
SITE# 2 (1980 - 1981)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

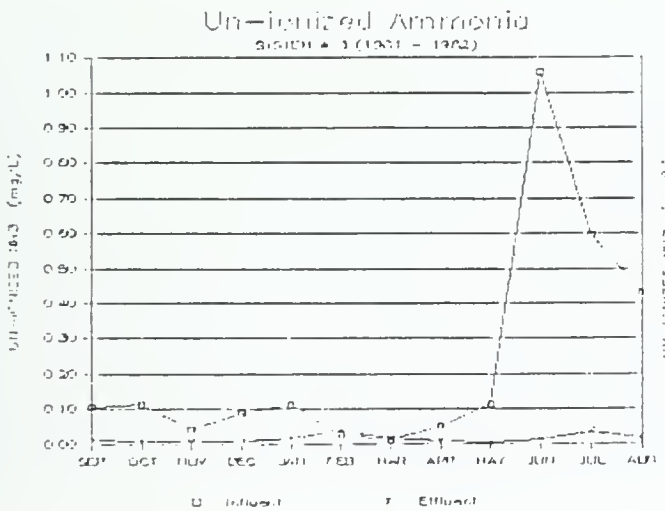

Un-ionized Ammonia
SITE# 2 (1982 - 1983)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

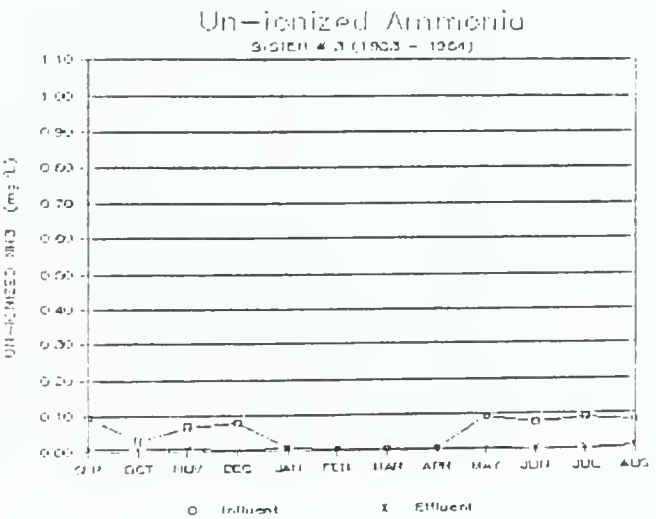

Un-ionized Ammonia
SITE# 2 (1981 - 1982)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

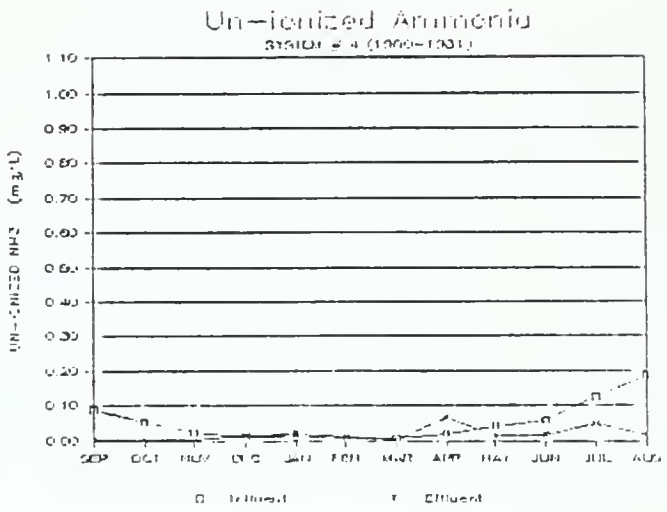

Un-ionized Ammonia
SITE# 2 (1983 - 1984)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

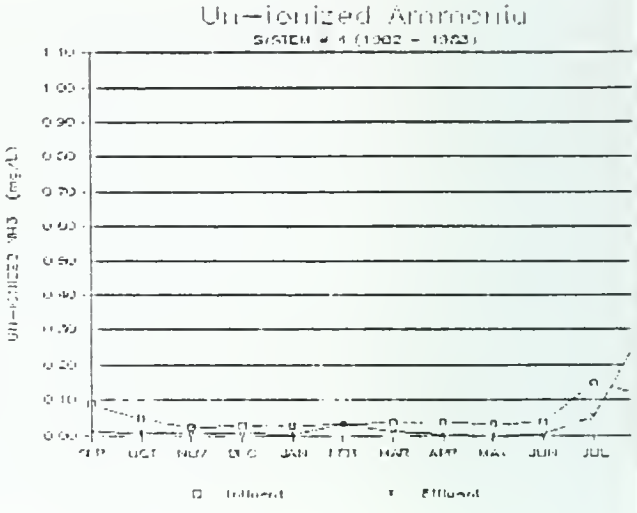

Un-ionized Ammonia
SISTEN # 3 (1900 - 1901)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent

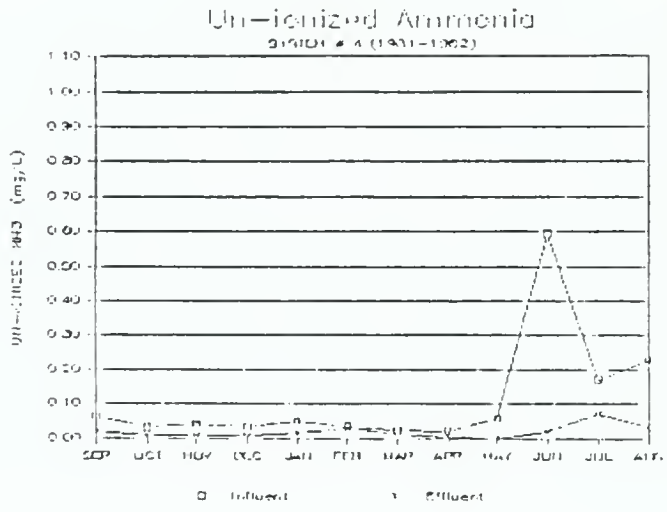

Un-ionized Ammonia
SISTEN # 3 (1902 - 1903)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent

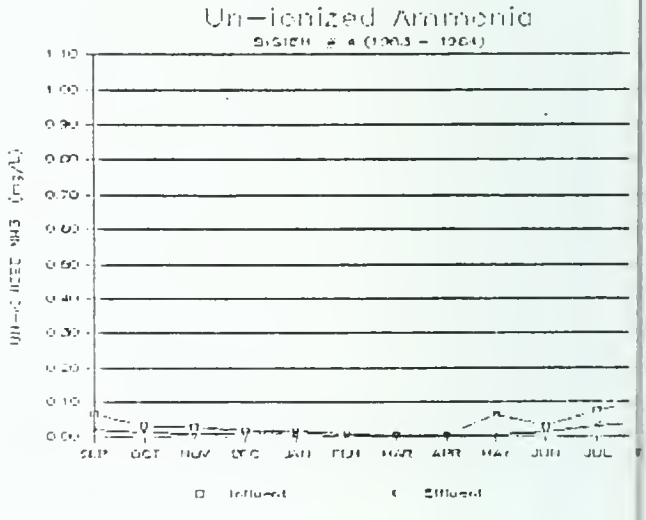

Un-ionized Ammonia
SISTEN # 3 (1901 - 1902)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent

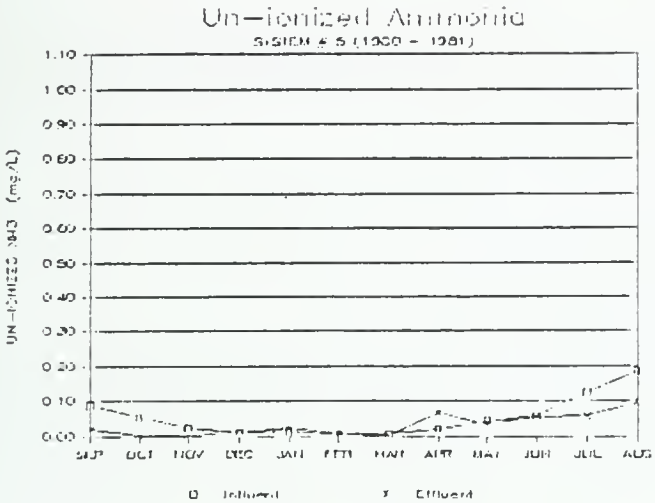

Un-ionized Ammonia
SISTEN # 3 (1903 - 1904)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
o Influent
x Effluent

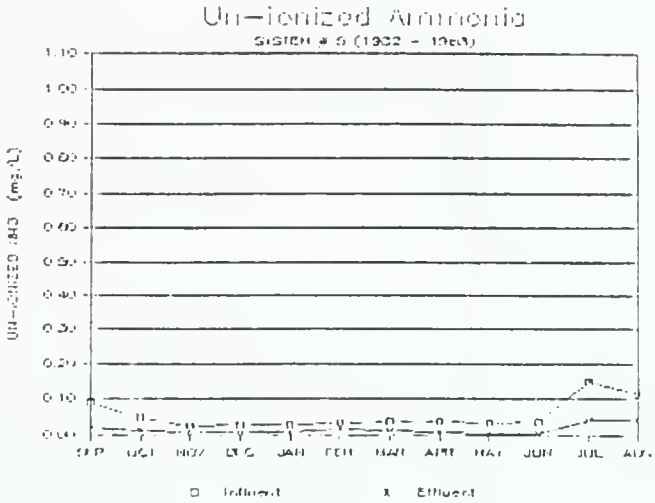

Un-ionized Ammonia
SYSTEM # 4 (1980-1981)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
Influent
Effluent

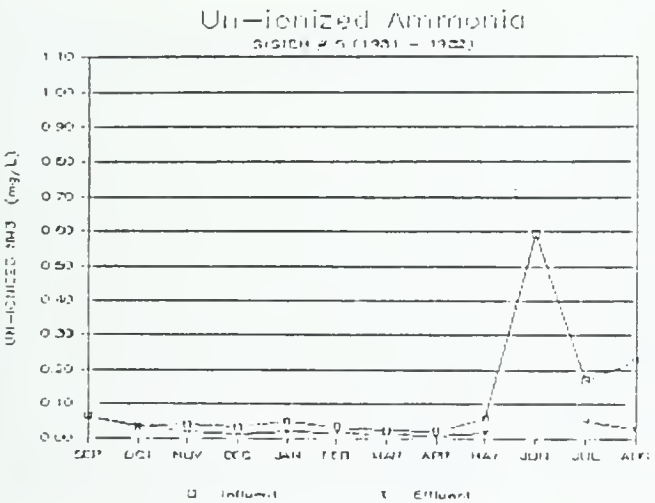

Un-ionized Ammonia
SYSTEM # 4 (1982 - 1983)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
Influent
Effluent

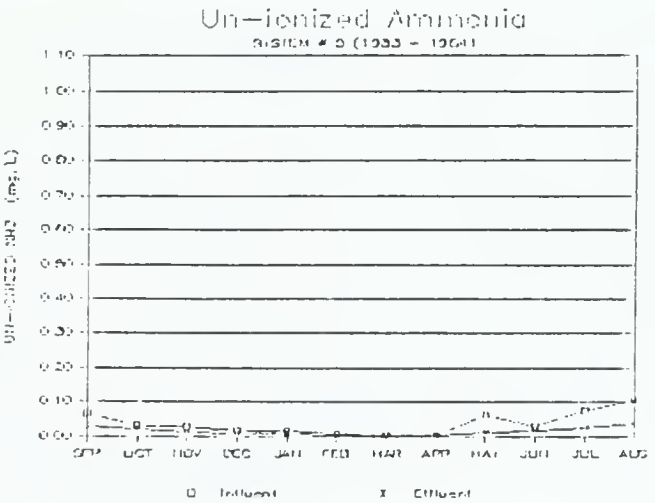

Un-ionized Ammonia
SYSTEM # 4 (1981-1982)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG
Influent
Effluent


Un-ionized Ammonia
SYSTEM # 4 (1983 - 1984)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
Influent
Effluent


Un-ionized Ammonia
SISIEN # 5 (1980 - 1981)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Un-ionized Ammonia
SISIEN # 5 (1982 - 1983)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Un-ionized Ammonia
SISIEN # 5 (1981 - 1982)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Un-ionized Ammonia
SISIEN # 5 (1983 - 1984)
UN-IONIZED NH3 (mg/L)
1.10
1.00
0.90
0.80
0.70
0.60
0.50
0.40
0.30
0.20
0.10
0.00
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

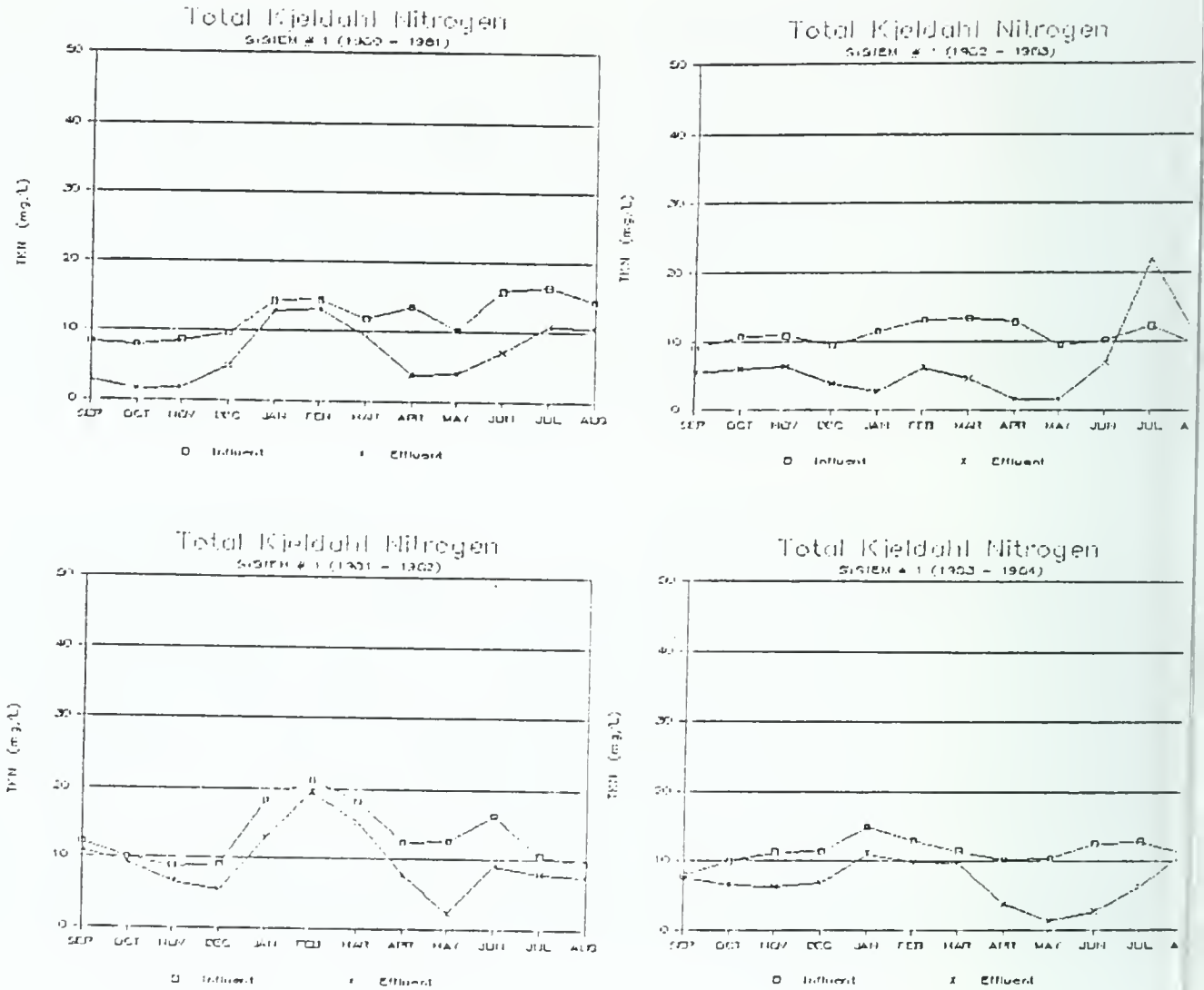
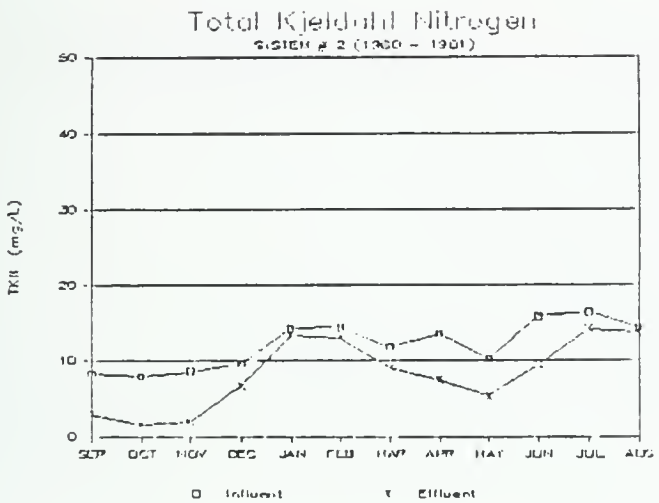
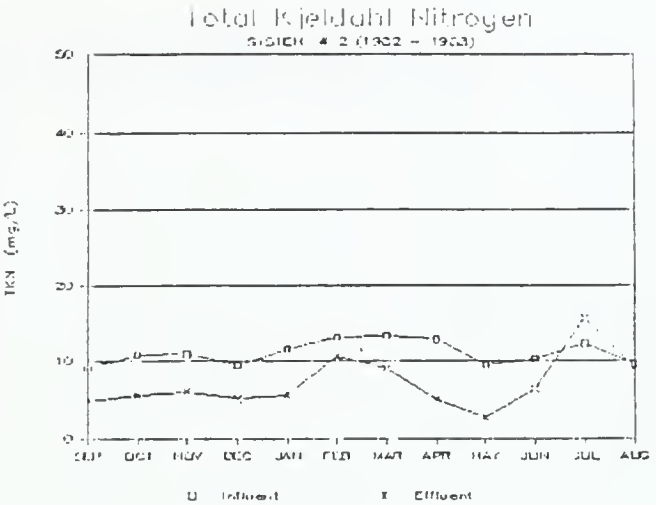
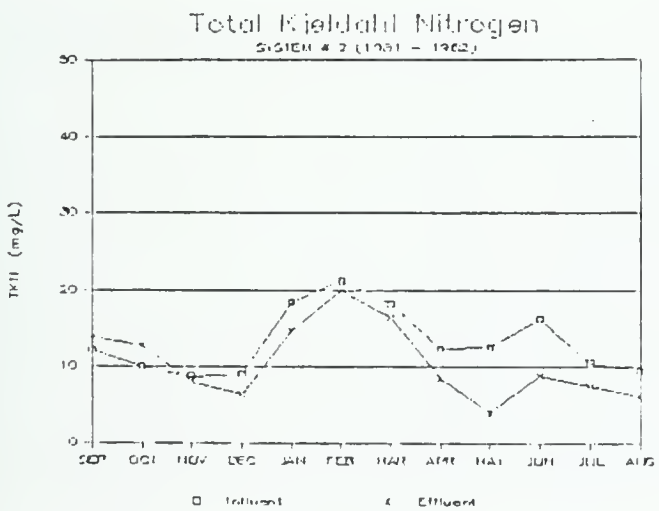
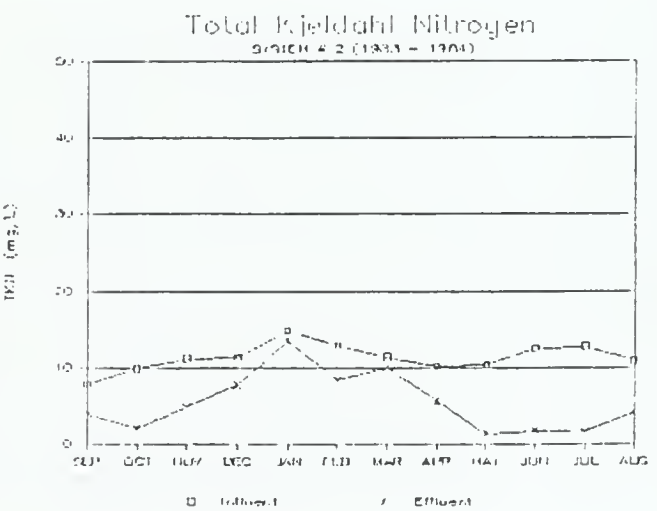


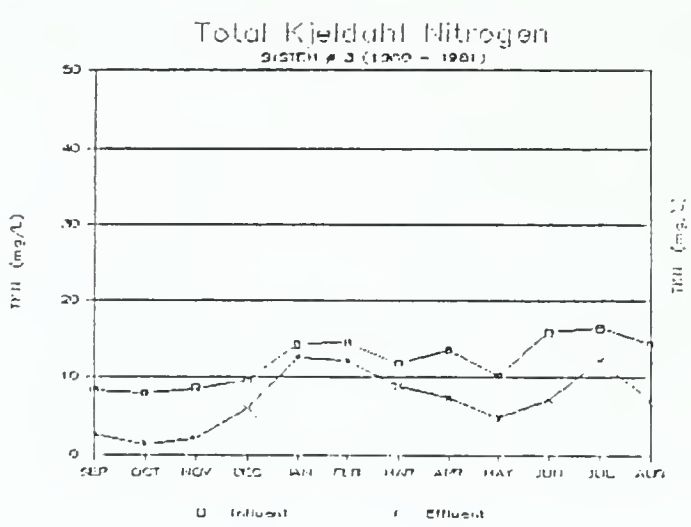
Figure A.18 Monthly average total Kjeldahl nitrogen concentrations in Systems 1-5 (4 years).

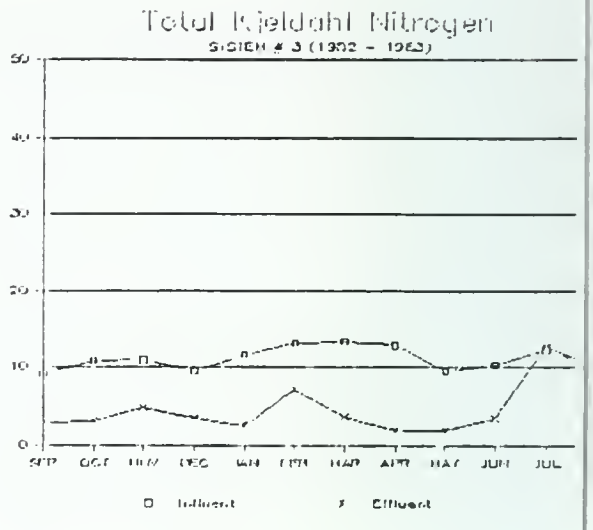

Total Kjeldahl Nitrogen
SISTEM # 2 (1980 - 1981)
TKN (mg/L)
0
10
20
30
40
50
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

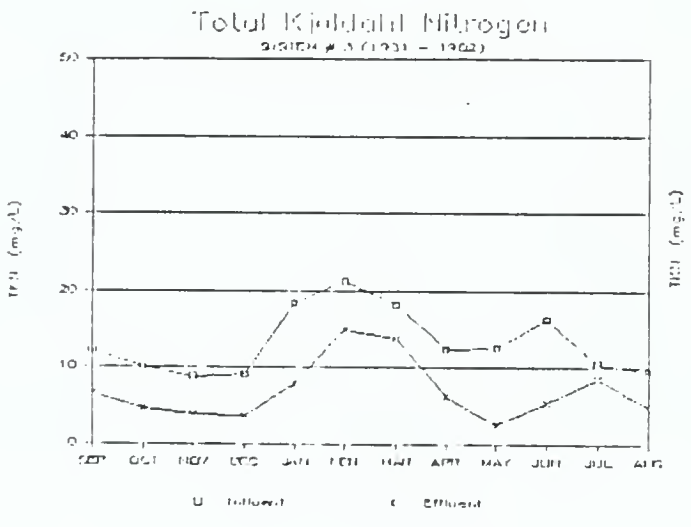

Total Kjeldahl Nitrogen
SISTEM # 2 (1982 - 1983)
TKN (mg/L)
0
10
20
30
40
50
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

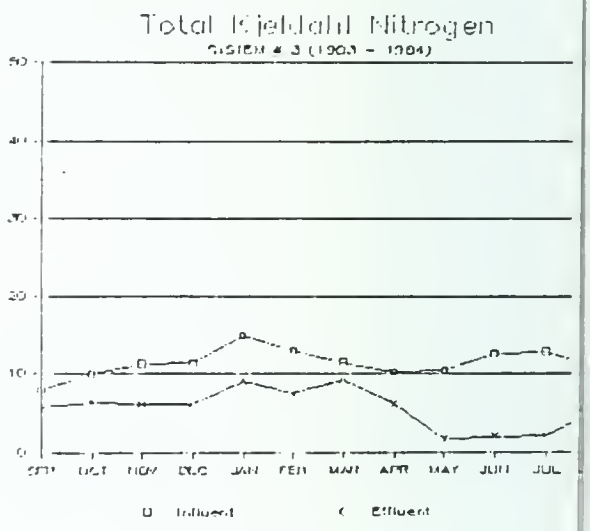

Total Kjeldahl Nitrogen
SISTEM # 2 (1981 - 1982)
TKN (mg/L)
0
10
20
30
40
50
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

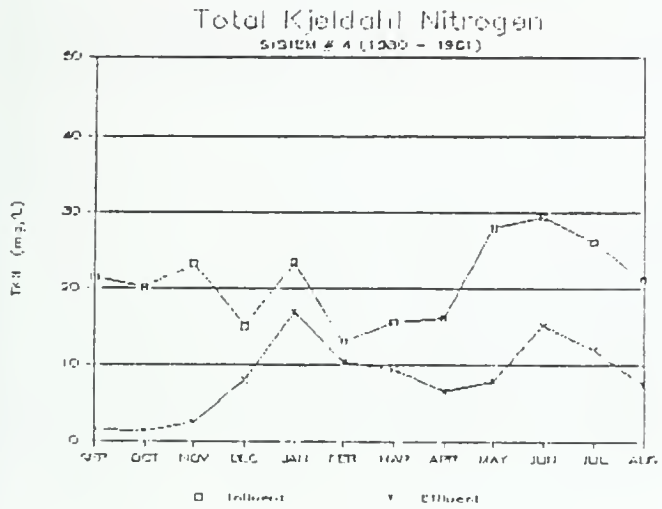

Total Kjeldahl Nitrogen
SISTEM # 2 (1983 - 1984)
TKN (mg/L)
0
10
20
30
40
50
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

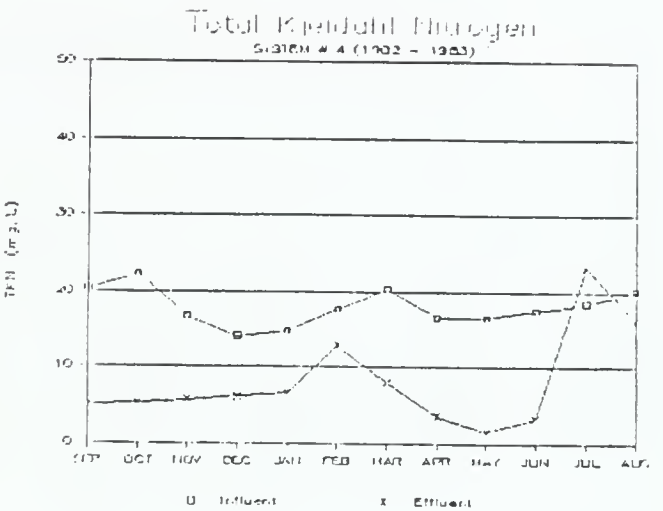

Total Kjeldahl Nitrogen
SYSTEM # 3 (1980 - 1981)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

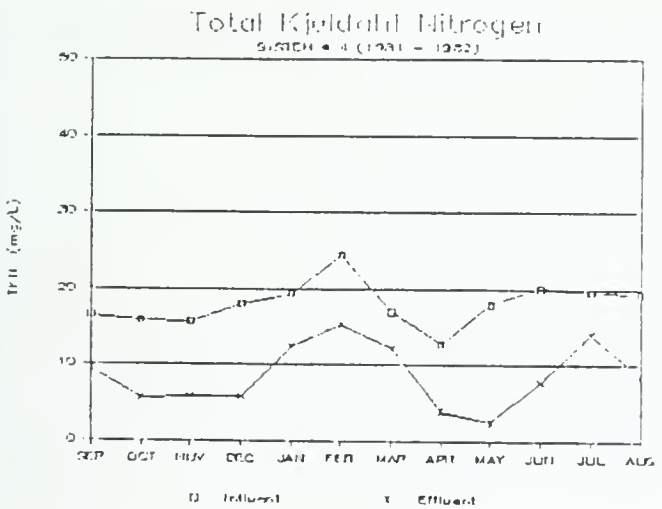

Total Kjeldahl Nitrogen
SYSTEM # 3 (1982 - 1983)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

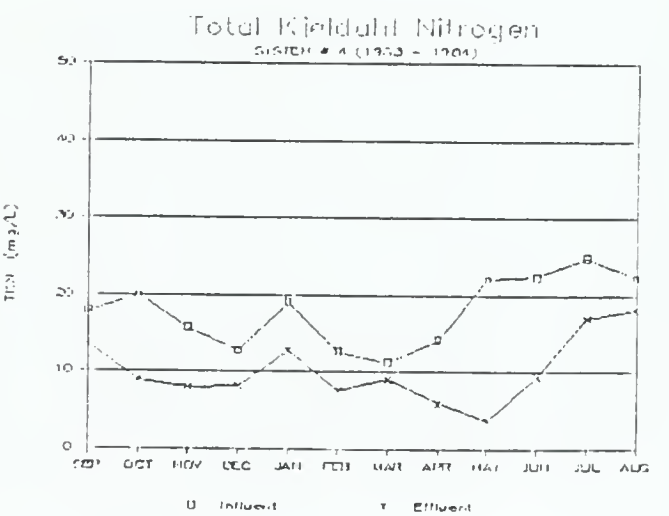

Total Kjeldahl Nitrogen
SYSTEM # 3 (1983 - 1984)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

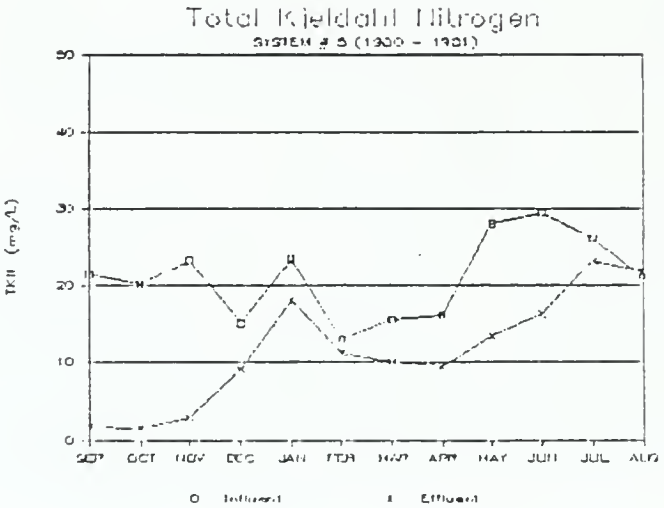

Total Kjeldahl Nitrogen
SYSTEM # 3 (1983 - 1984)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
Influent
Effluent

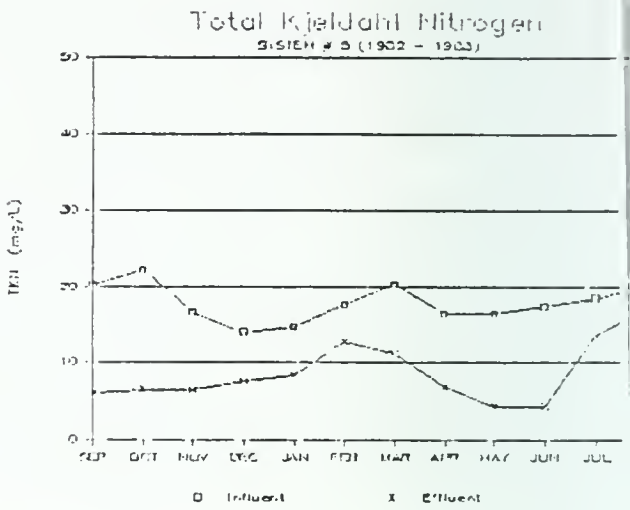

Total Kjeldahl Nitrogen
SISTEM # 4 (1980 - 1981)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
□ Influent
x Effluent

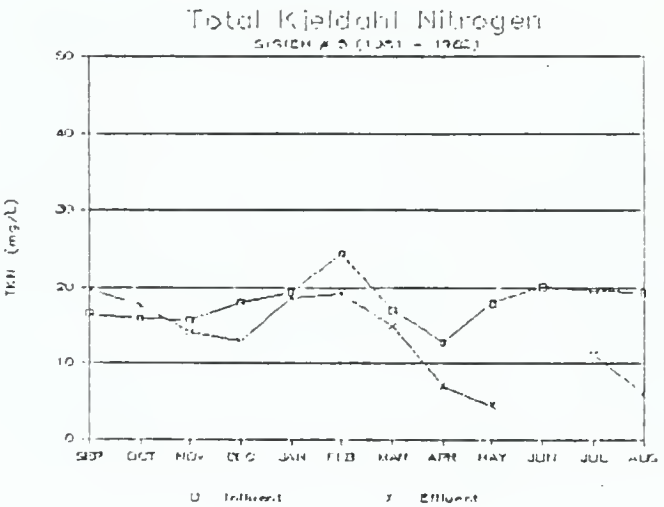

Total Kjeldahl Nitrogen
SISTEM # 4 (1982 - 1983)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
□ Influent
x Effluent

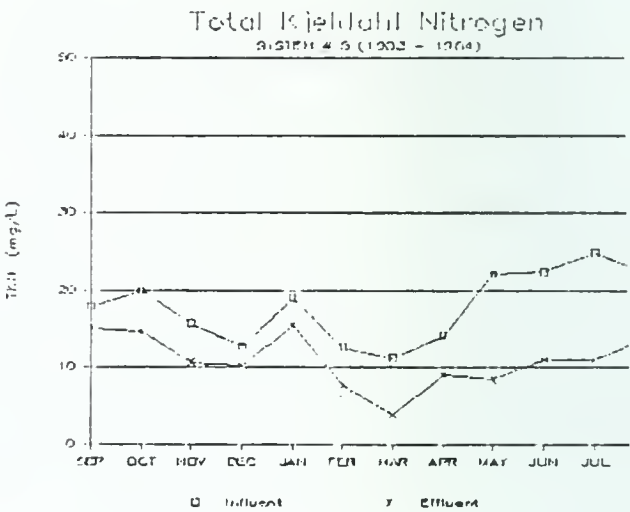

Total Kjeldahl Nitrogen
SISTEM # 4 (1981 - 1982)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
□ Influent
x Effluent


Total Kjeldahl Nitrogen
SISTEM # 4 (1983 - 1984)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
□ Influent
x Effluent


Total Kjeldahl Nitrogen
SYSTEM # 5 (1990 - 1991)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Total Kjeldahl Nitrogen
SYSTEM # 5 (1992 - 1993)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent


Total Kjeldahl Nitrogen
SYSTEM # 5 (1991 - 1992)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent


Total Kjeldahl Nitrogen
SYSTEM # 5 (1993 - 1994)
TKN (mg/L)
50
40
30
20
10
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

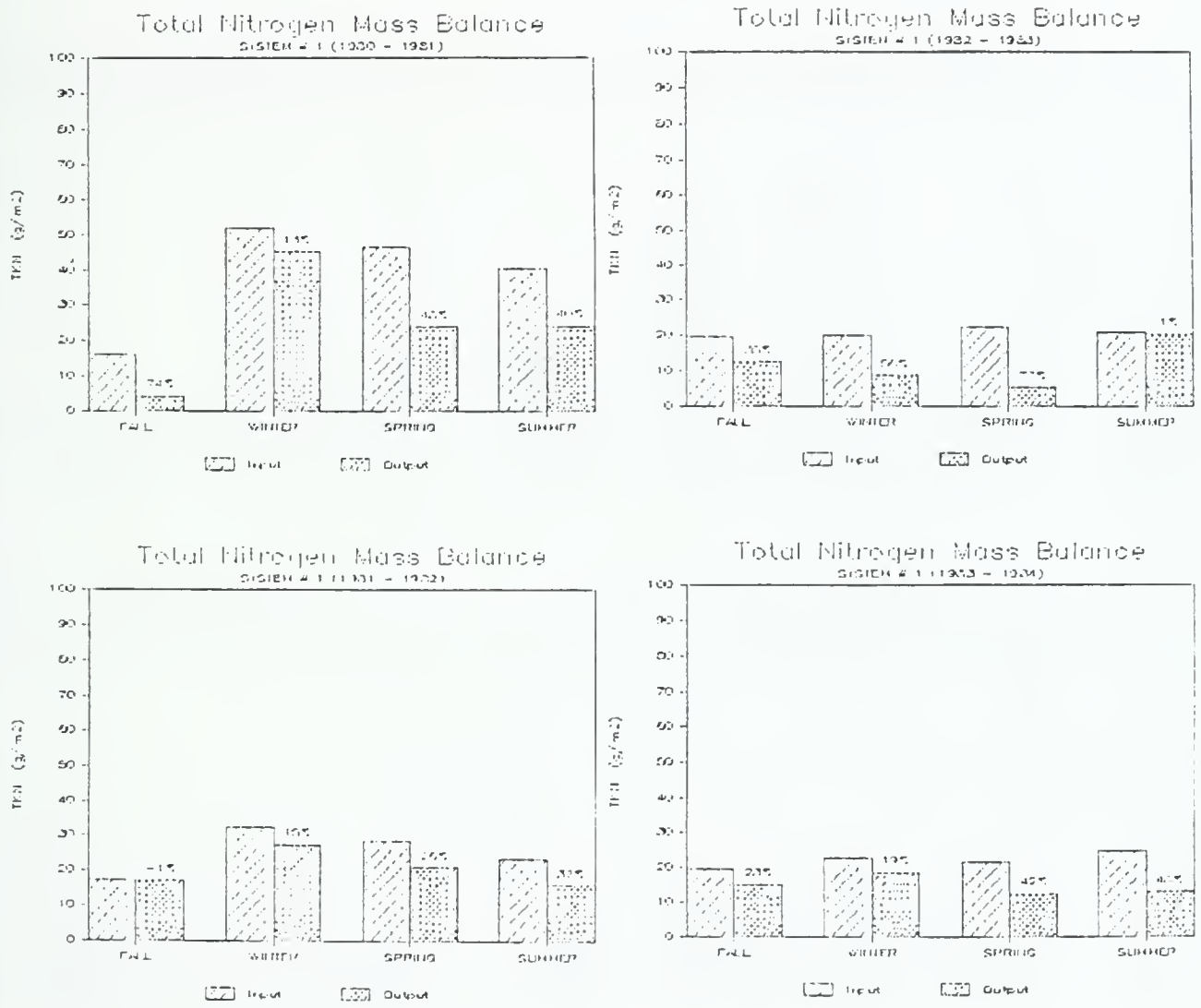
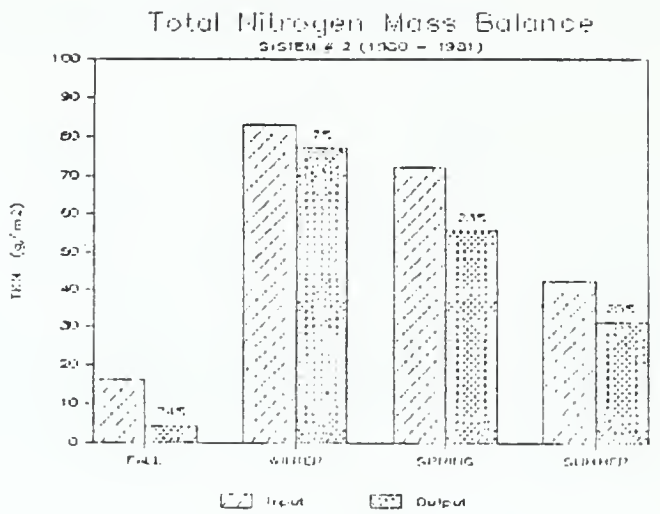
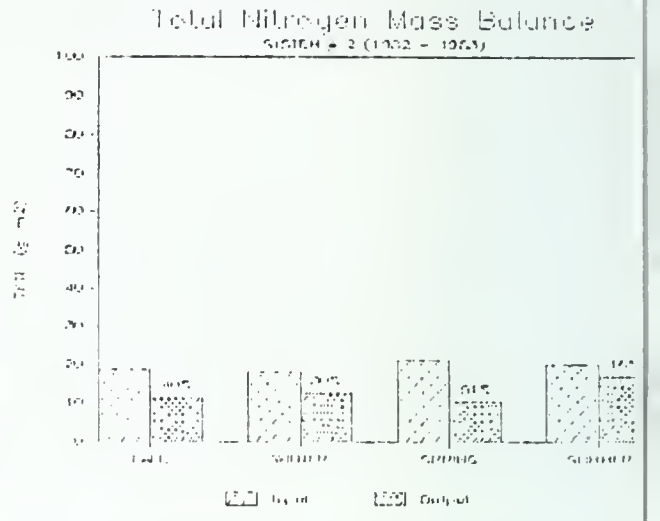
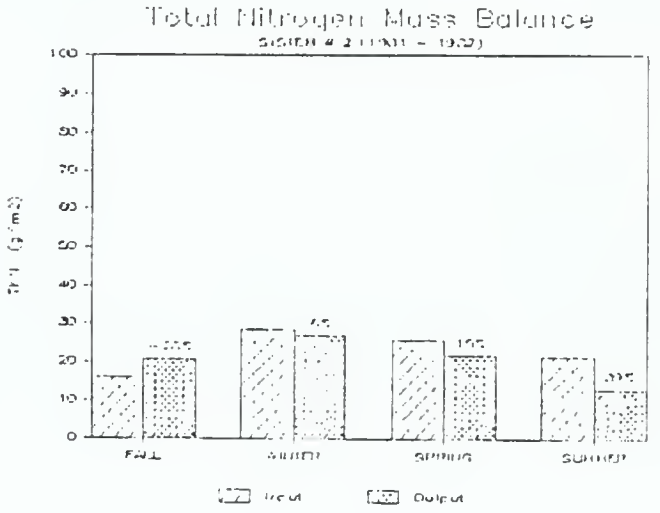
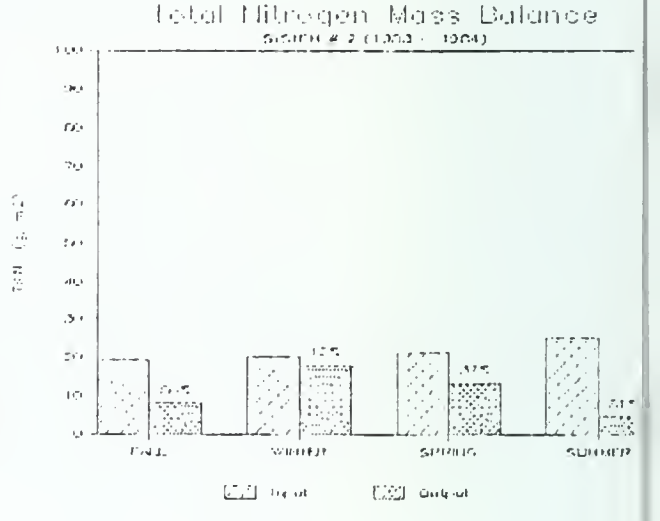


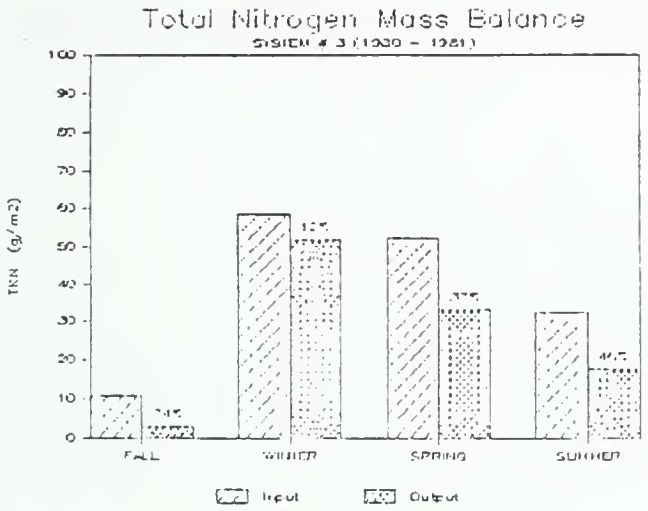
Figure A.19 Seasonal total Kjeldahl nitrogen mass balances and removal efficiencies in Systems 1-5 (4 years).

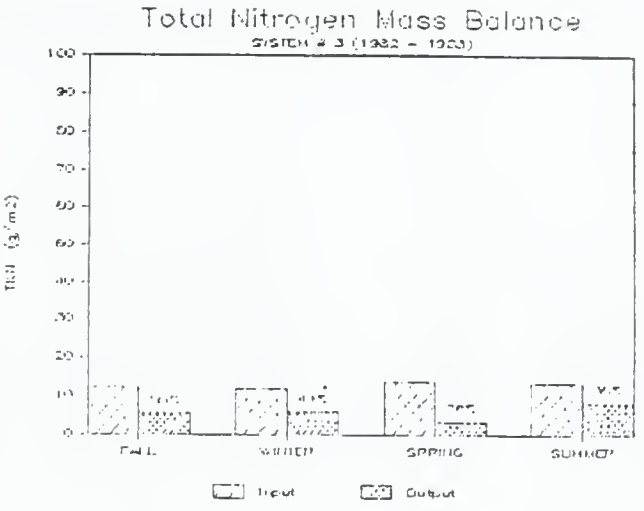

Total Nitrogen Mass Balance
SYSTEM # 2 (1980 - 1981)
TEM (g/m2)
100
90
80
70
60
50
40
30
20
10
0
FALL
WINTER
SPRING
SUMMER
Input
Output

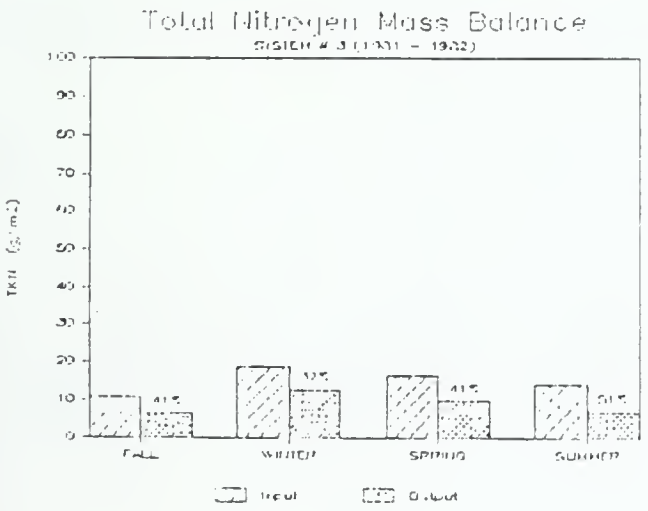

Total Nitrogen Mass Balance
SYSTEM # 2 (1982 - 1983)
TEM (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

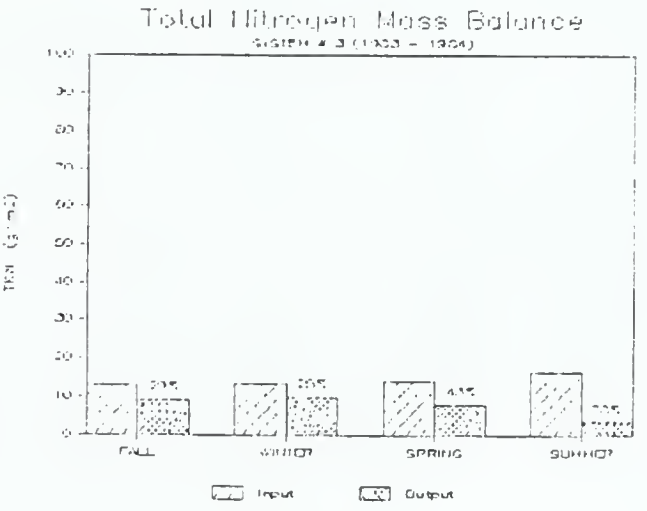

Total Nitrogen Mass Balance
SYSTEM # 2 (1981 - 1982)
TEM (g/m2)
100
90
80
70
60
50
40
30
20
10
0
FALL
WINTER
SPRING
SUMMER
Input
Output

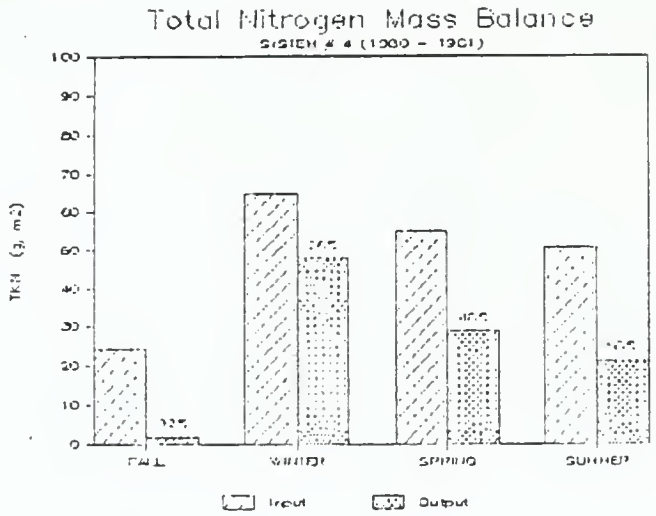

Total Nitrogen Mass Balance
SYSTEM # 2 (1983 - 1984)
TEM (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

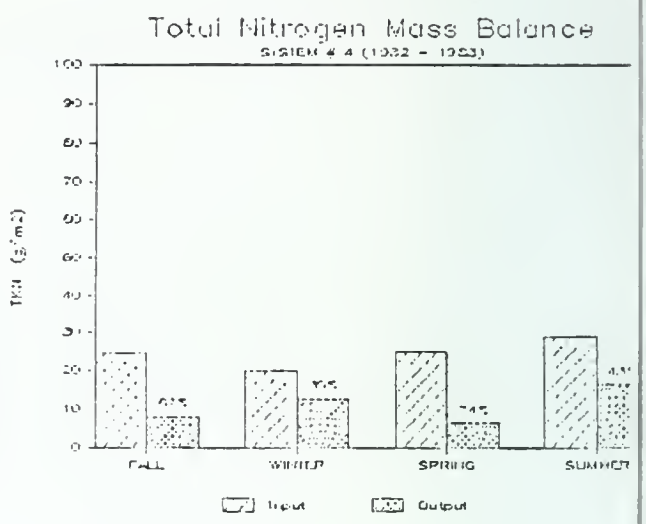

Total Nitrogen Mass Balance
SYSTEM # 3 (1980 - 1981)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

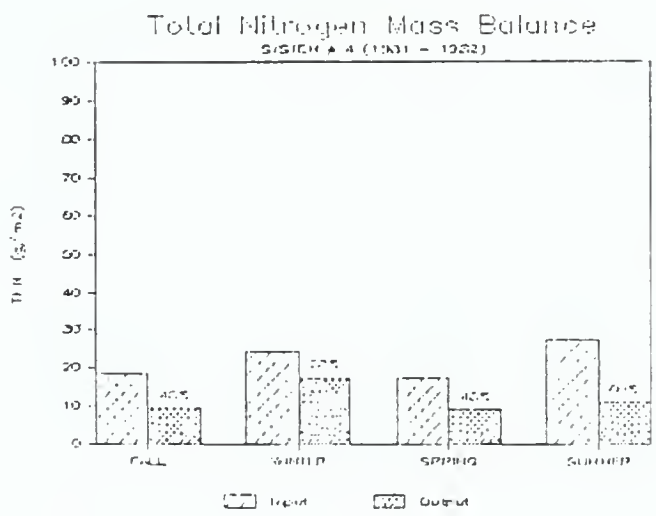

Total Nitrogen Mass Balance
SYSTEM # 3 (1982 - 1983)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

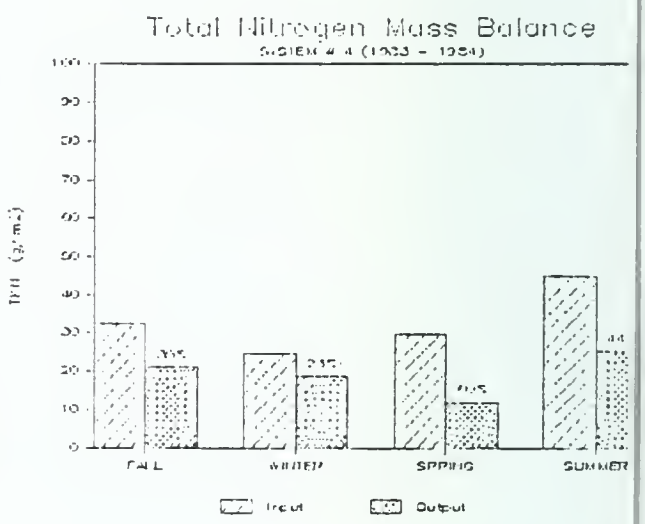

Total Nitrogen Mass Balance
SYSTEM # 3 (1981 - 1982)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

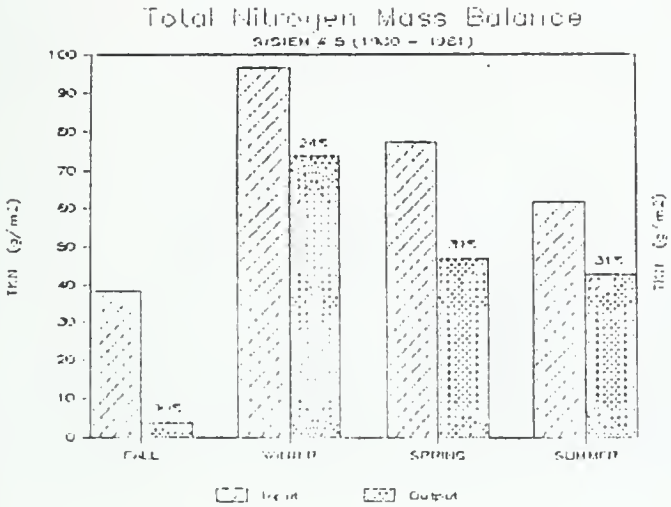

Total Nitrogen Mass Balance
SYSTEM # 3 (1983 - 1984)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

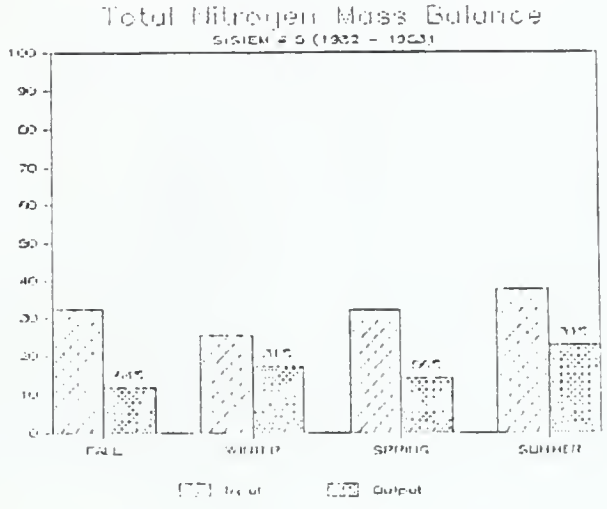

Total Nitrogen Mass Balance
SISTEM # 4 (1980 - 1981)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

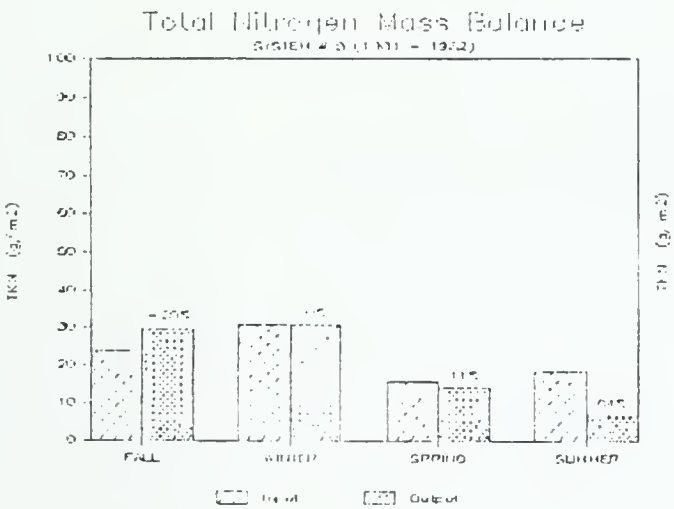

Total Nitrogen Mass Balance
SISTEM # 4 (1982 - 1983)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

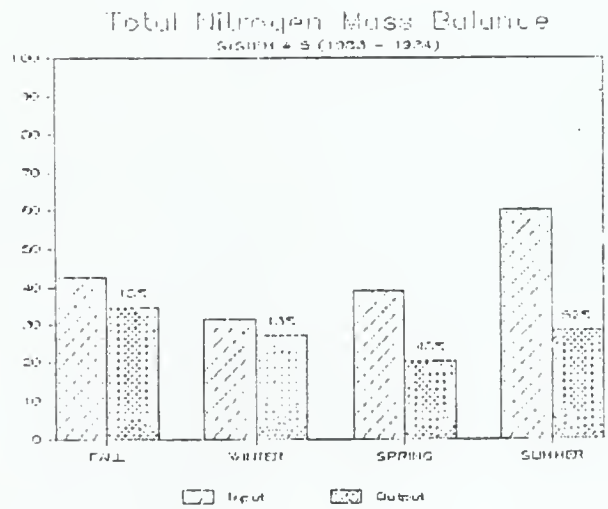

Total Nitrogen Mass Balance
SISTEM # 4 (1981 - 1982)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

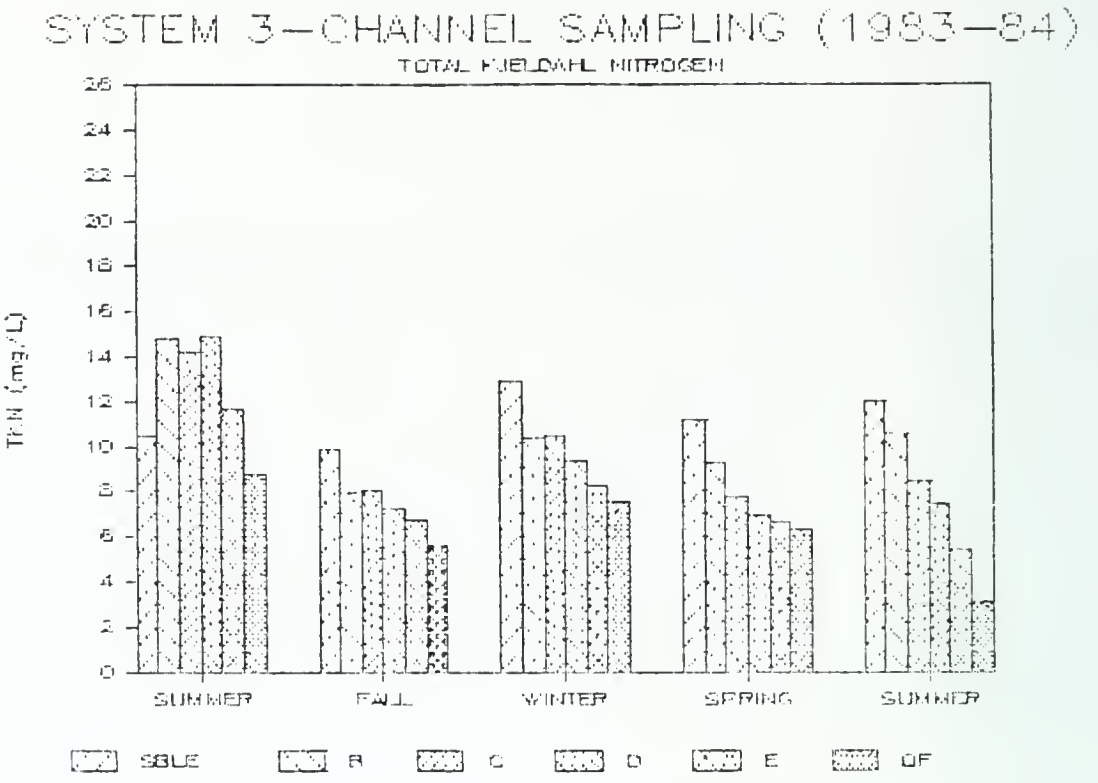

Total Nitrogen Mass Balance
SISTEM # 4 (1983 - 1984)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Nitrogen Mass Balance
SISIEN # 5 (1980 - 1981)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Nitrogen Mass Balance
SISIEN # 5 (1982 - 1983)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Nitrogen Mass Balance
SISIEN # 5 (1981 - 1982)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output


Total Nitrogen Mass Balance
SISIEN # 5 (1983 - 1984)
TKN (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output



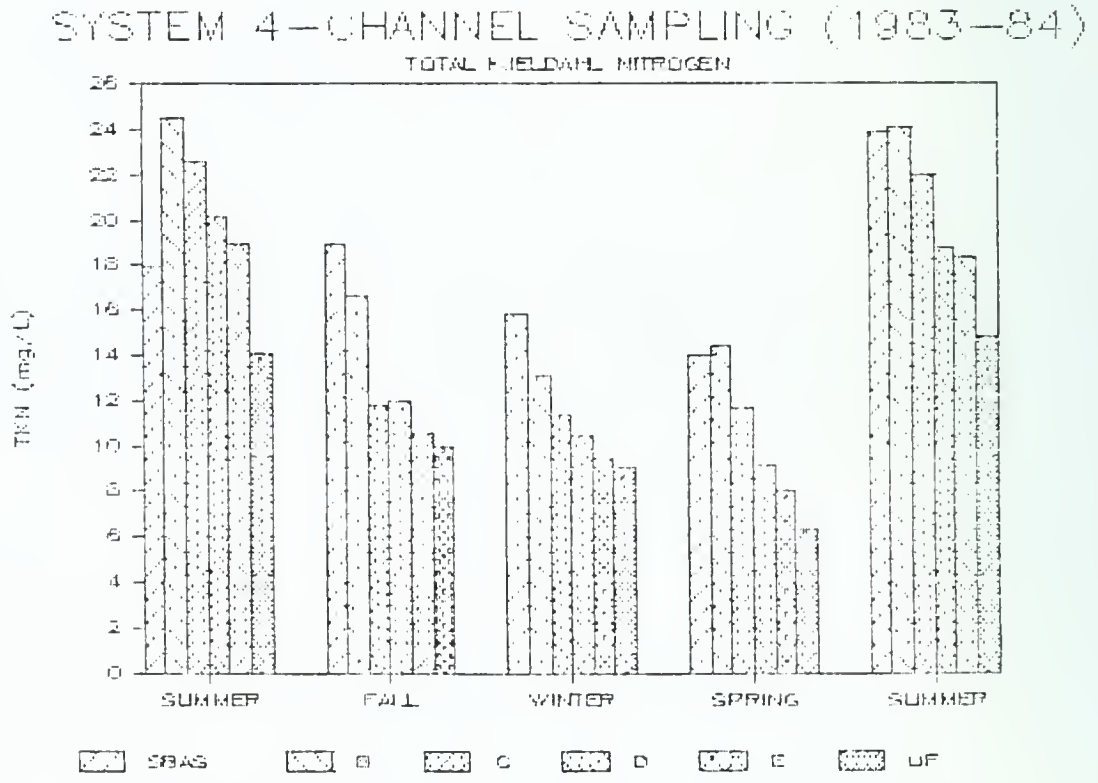


Figure A.20 Seasonal total Kjeldahl nitrogen concentrations in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of Systems 3 and 4 (1983-4).

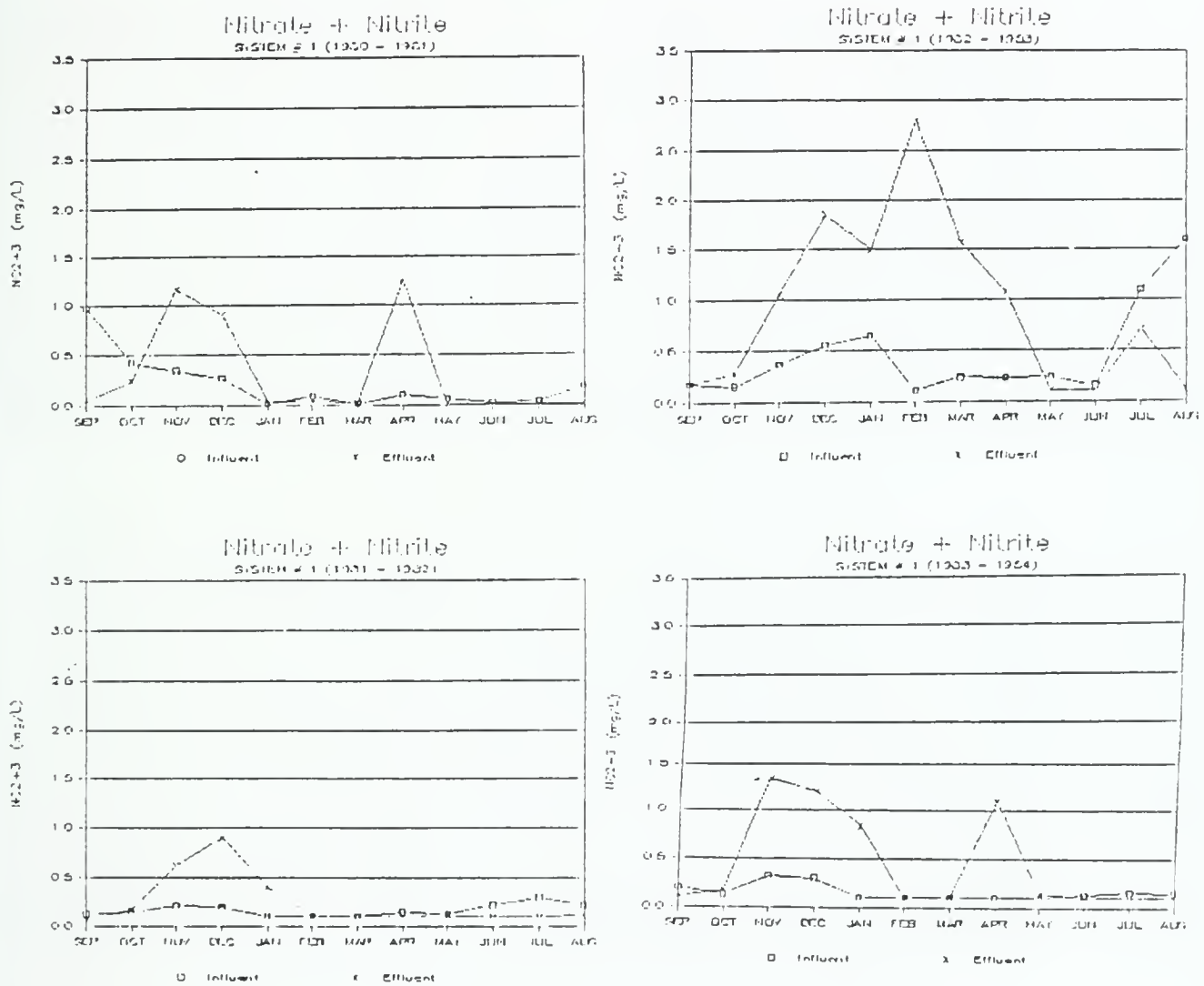
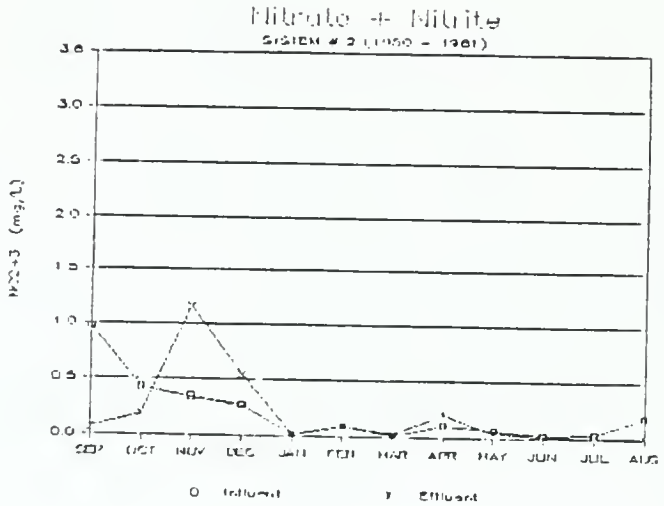
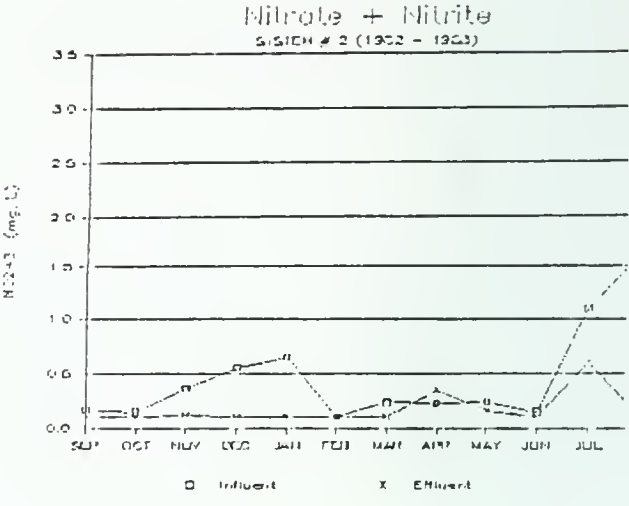
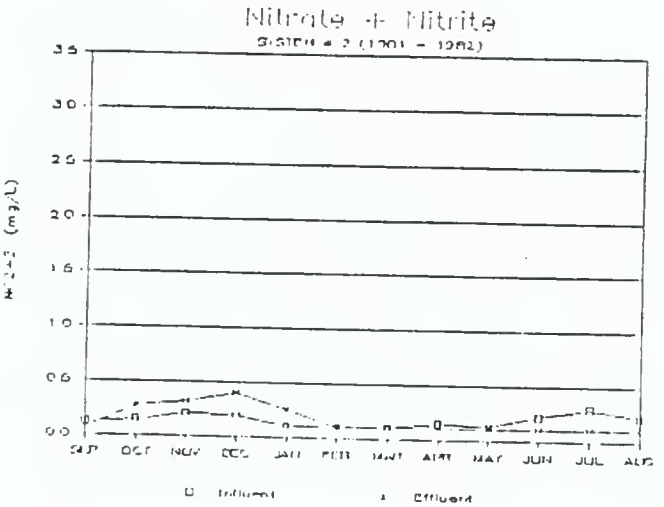
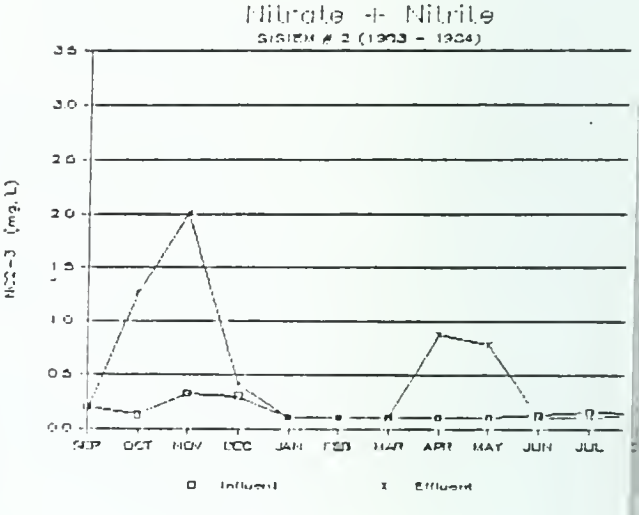

Nitrate + Nitrite
SYSTEM # 1 (1980 - 1981)
Nitrate + Nitrite
SYSTEM # 1 (1982 - 1983)
Nitrate + Nitrite
SYSTEM # 1 (1983 - 1984)
Nitrate + Nitrite
SYSTEM # 1 (1983 - 1984)
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

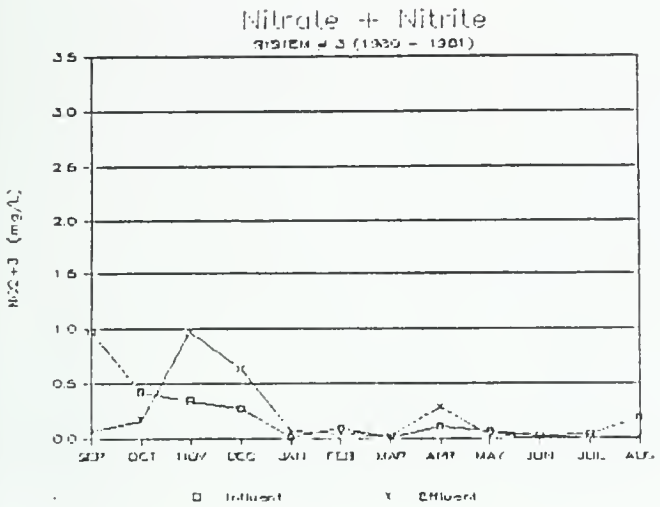
Figure A.21 Monthly average nitrate plus nitrite concentrations in Systems 1-5 (4 years). (Values < detection limit, i.e. 0.01 mg/L in year 1 and 0.1 mg/L in years 2-4, recorded as at limit)

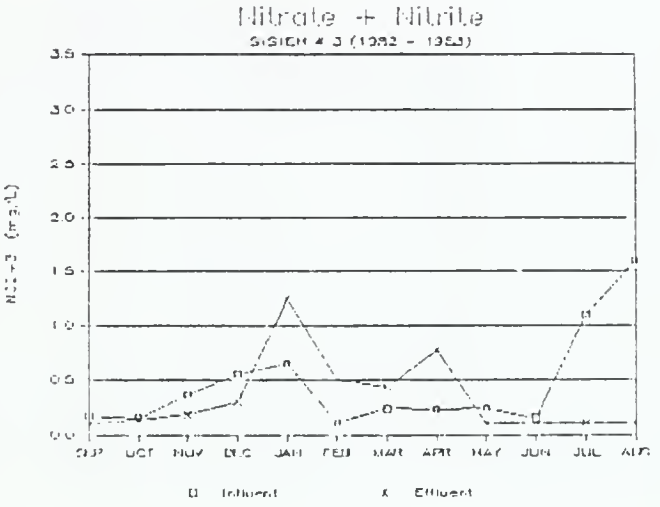

Nitrate + Nitrite
SISIEN # 2 (1980 - 1981)
NO3-3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

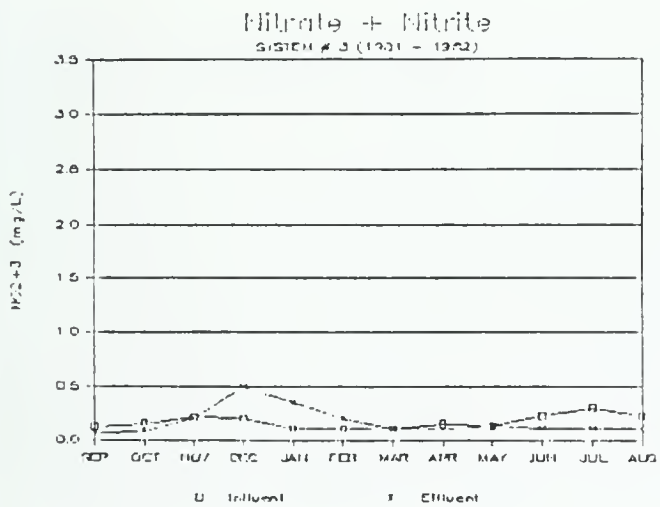

Nitrate + Nitrite
SISIEN # 2 (1982 - 1983)
NO3-3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

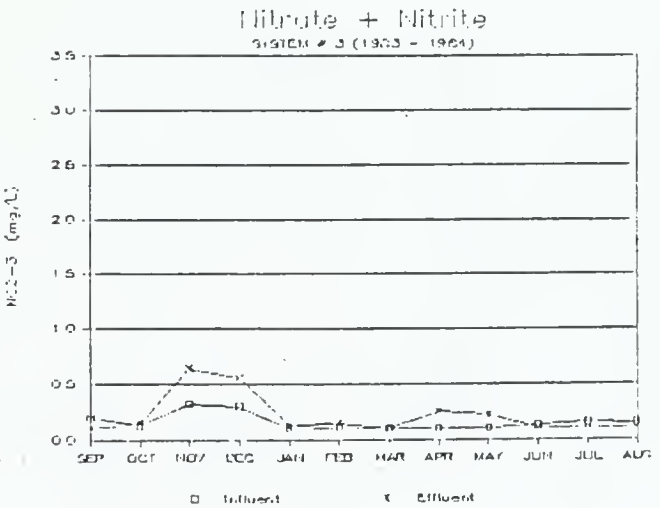

Nitrate + Nitrite
SISIEN # 2 (1981 - 1982)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
X Effluent

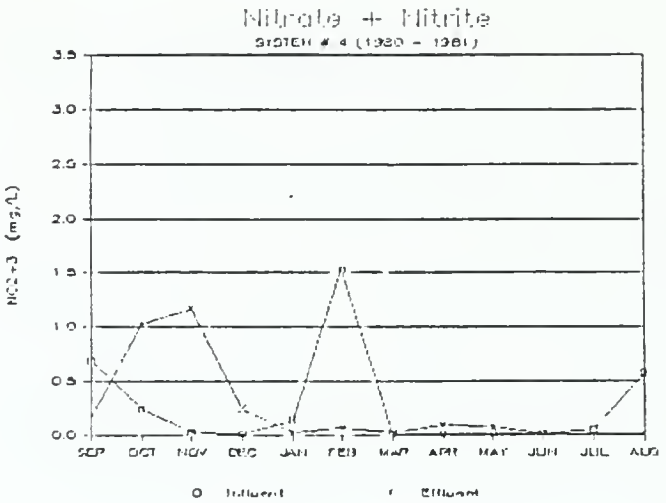

Nitrate + Nitrite
SISIEN # 2 (1983 - 1984)
NO3-3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

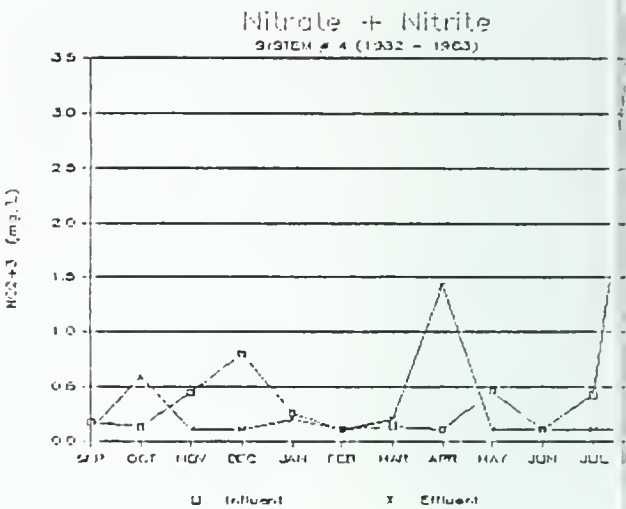

Nitrate + Nitrite
SYSTEM # 3 (1980 - 1981)
NO3+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

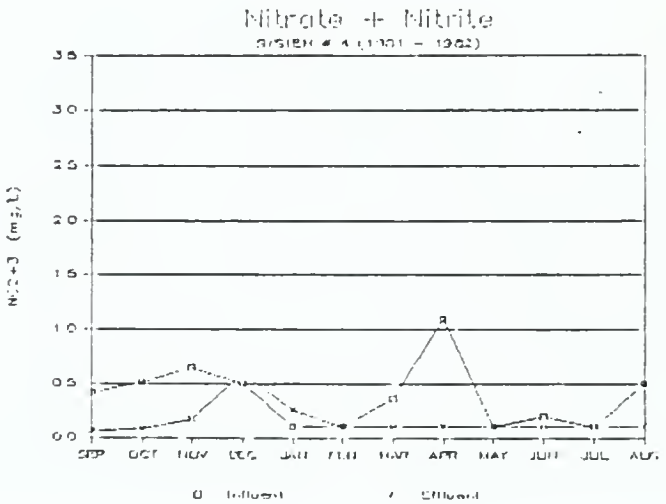

Nitrate + Nitrite
SYSTEM # 3 (1982 - 1983)
NO3+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

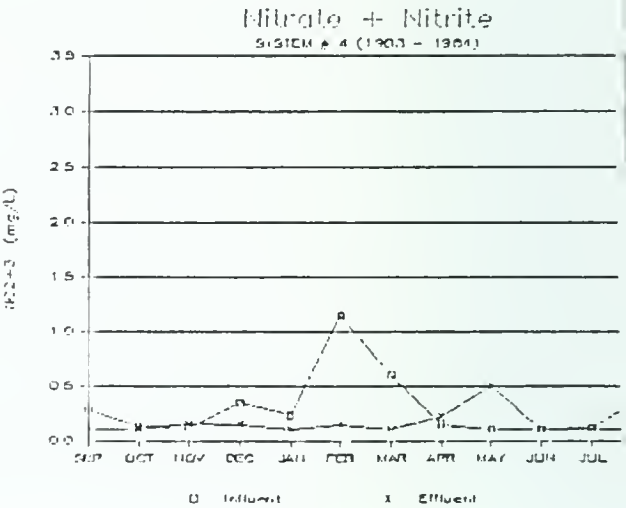

Nitrate + Nitrite
SYSTEM # 3 (1981 - 1982)
NO3+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

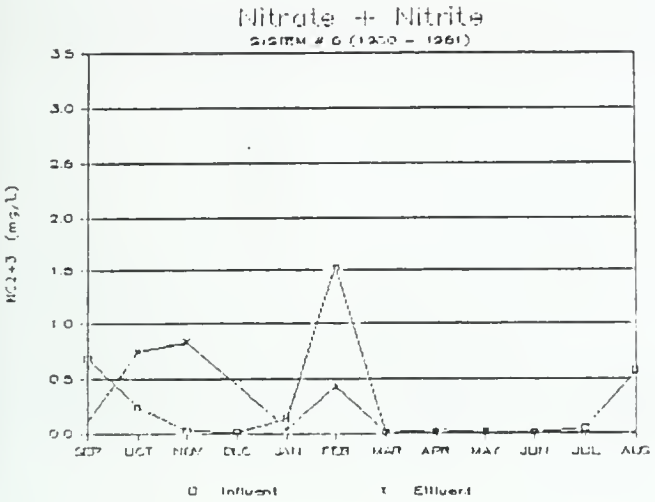

Nitrate + Nitrite
SYSTEM # 3 (1983 - 1984)
NO3+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

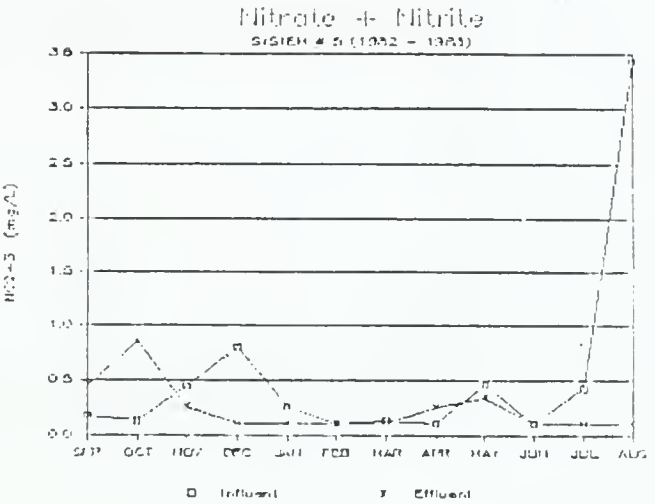

Nitrate + Nitrite
SYSTEM # 4 (1980 - 1981)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
Effluent

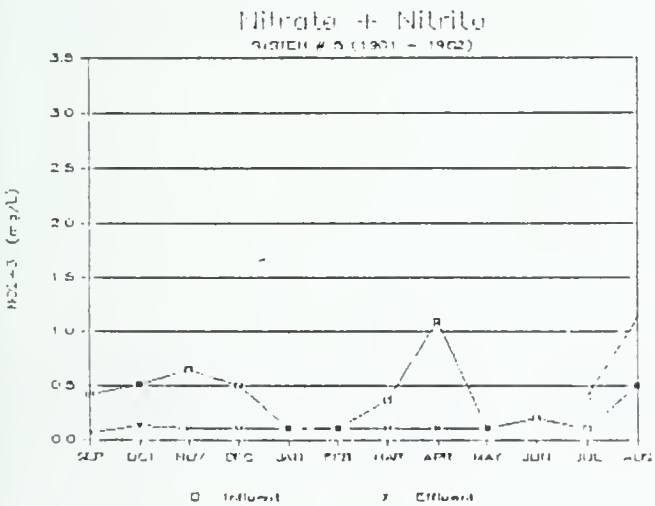

Nitrate + Nitrite
SYSTEM # 4 (1982 - 1983)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent

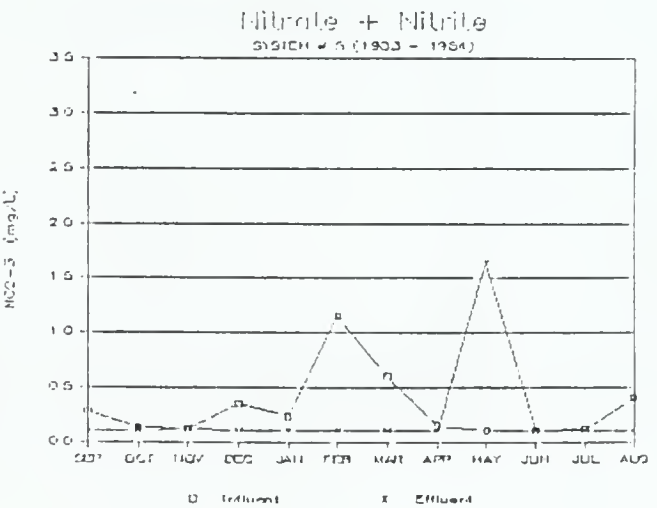

Nitrate + Nitrite
SYSTEM # 4 (1981 - 1982)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
O Influent
Effluent


Nitrate + Nitrite
SYSTEM # 4 (1983 - 1984)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL
O Influent
X Effluent


Nitrate + Nitrite
SYSTEM # 5 (1980 - 1981)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Nitrate + Nitrite
SYSTEM # 5 (1982 - 1983)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Nitrate + Nitrite
SYSTEM # 5 (1981 - 1982)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent


Nitrate + Nitrite
SYSTEM # 5 (1983 - 1984)
NO2+3 (mg/L)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
Influent
Effluent

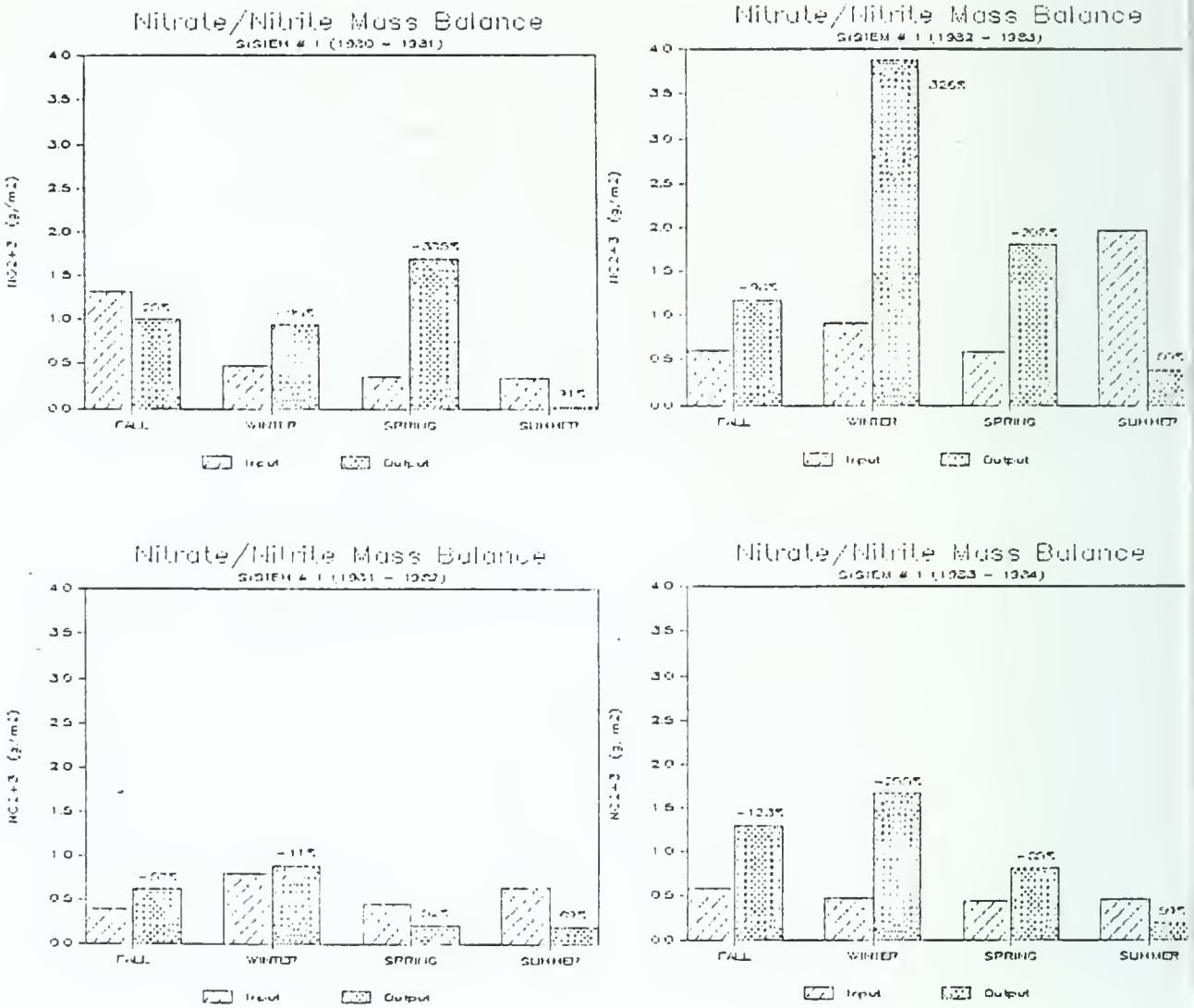
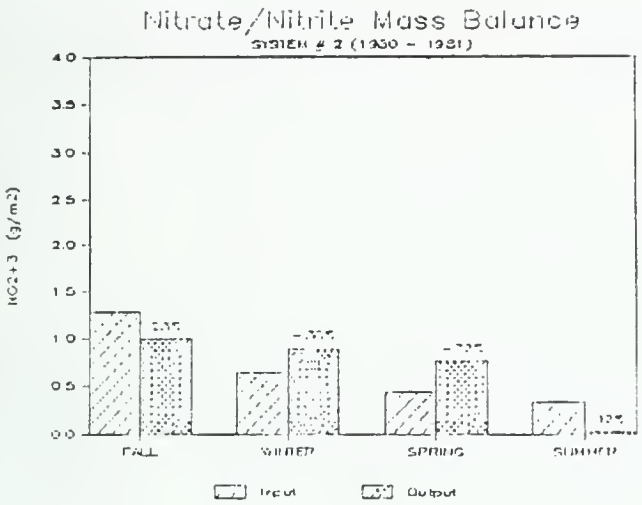
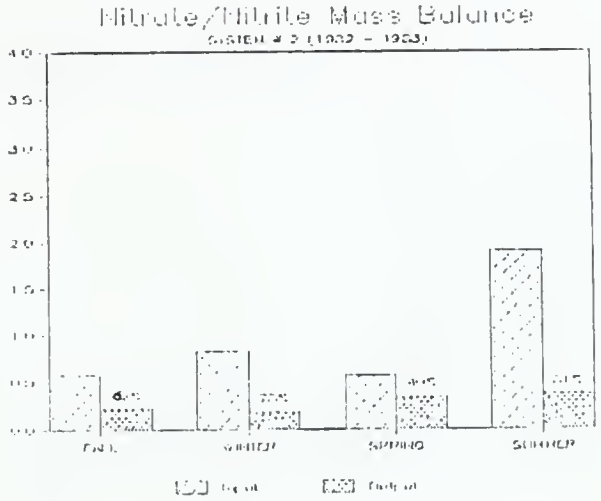
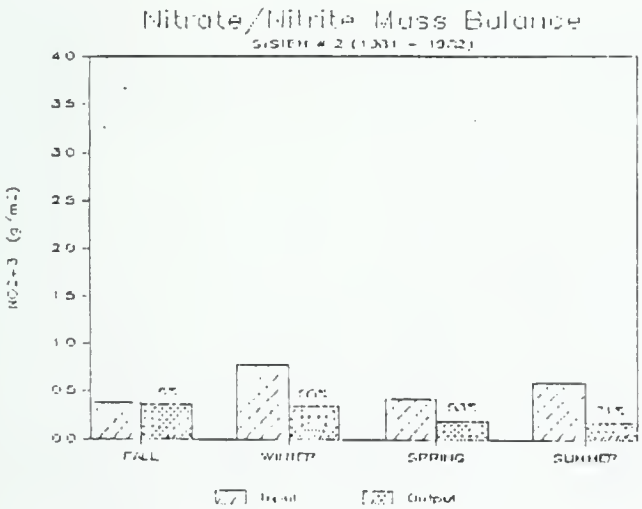
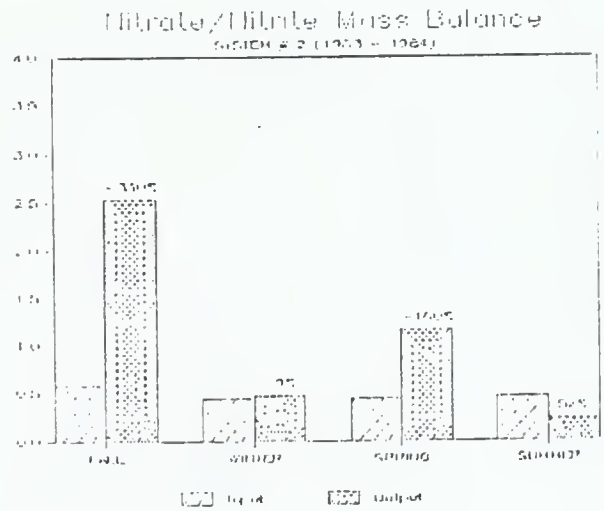


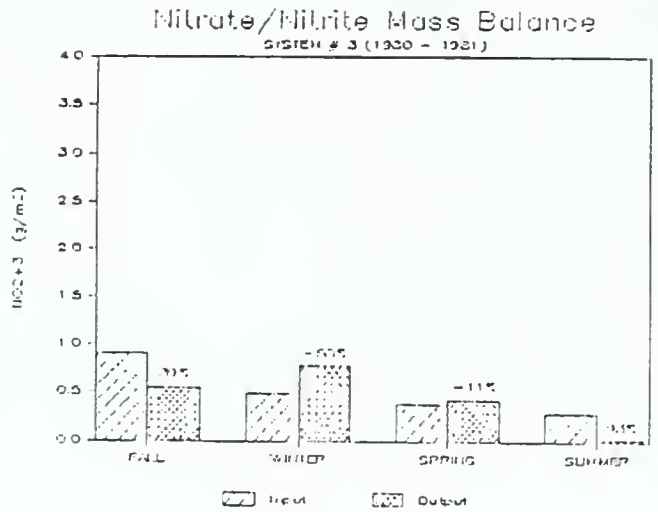
Figure A.22 Seasonal nitrate/nitrite mass balances and removal efficiencies in Systems 1-5 (4 years).

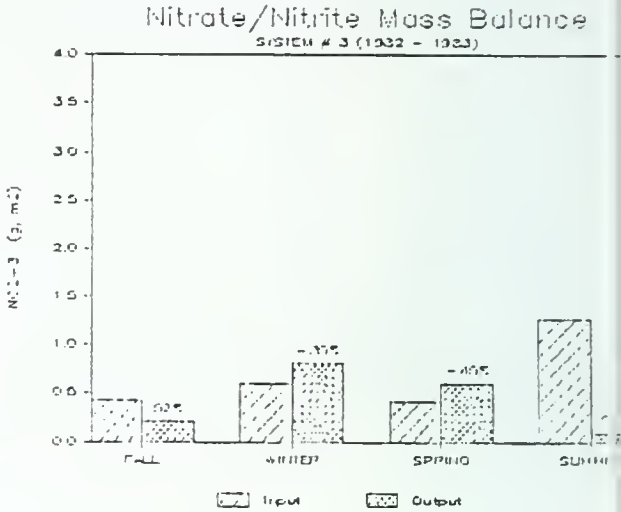

Nitrate/Nitrite Mass Balance
SISTEM # 2 (1980 - 1981)
NO2+3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

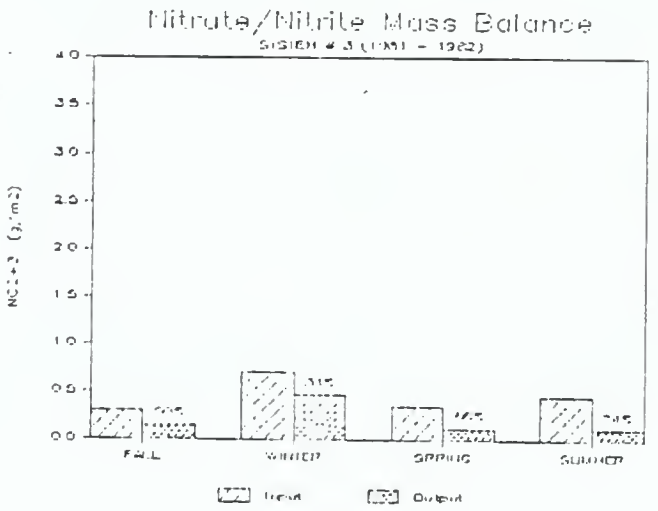

Nitrate/Nitrite Mass Balance
NO2+3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

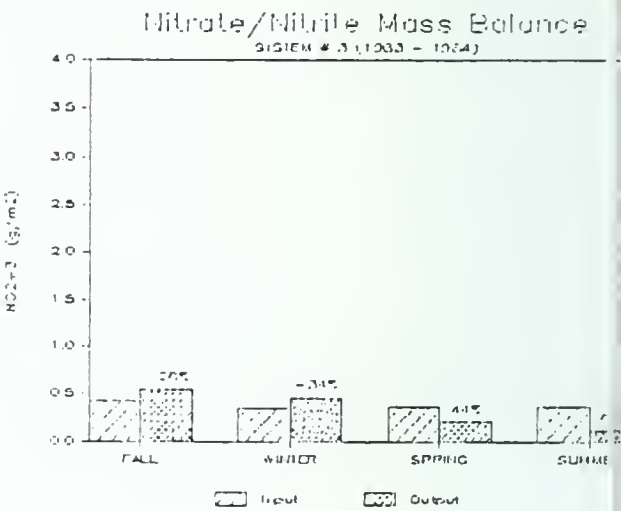

Nitrate/Nitrite Mass Balance
SISTEM # 2 (1981 - 1982)
NO2+3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

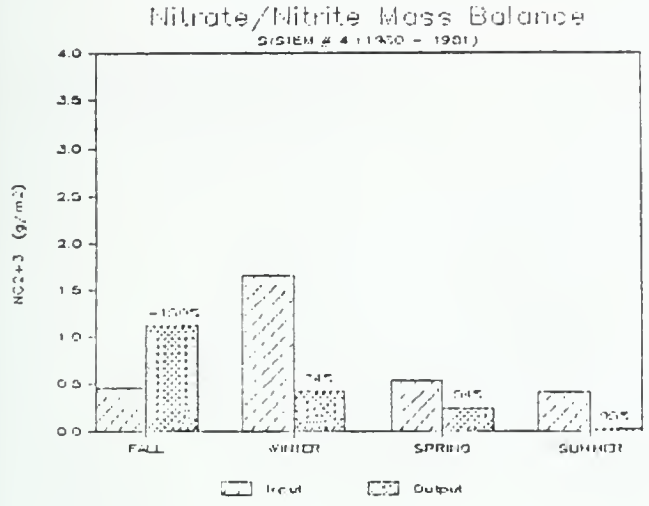

Nitrate/Nitrite Mass Balance
SISTEM # 2 (1983 - 1984)
NO2+3 (g/m2)
FALL
WINTER
SPRING
SUMMER
Input
Output

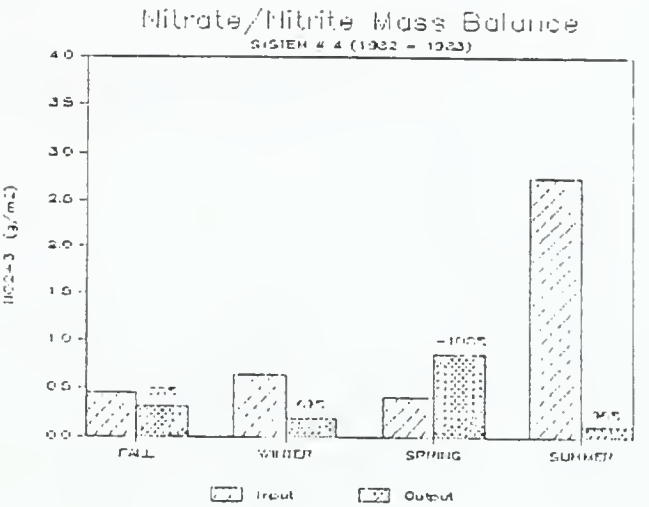

Nitrate/Nitrite Mass Balance
SYSTEM # 3 (1980 - 1981)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

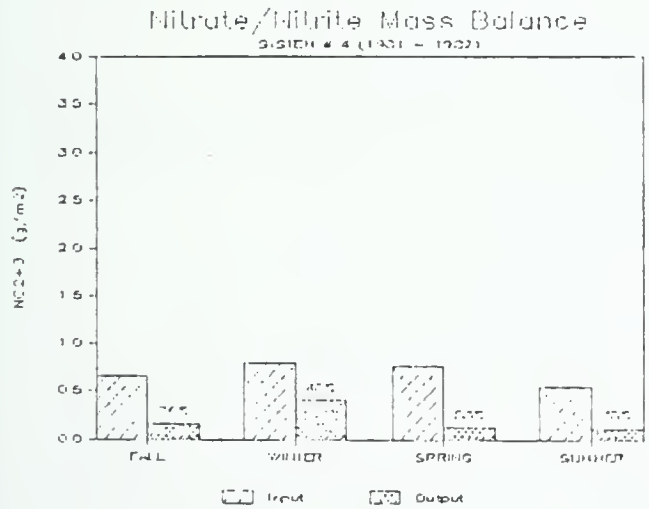

Nitrate/Nitrite Mass Balance
SYSTEM # 3 (1982 - 1983)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
Input
Output

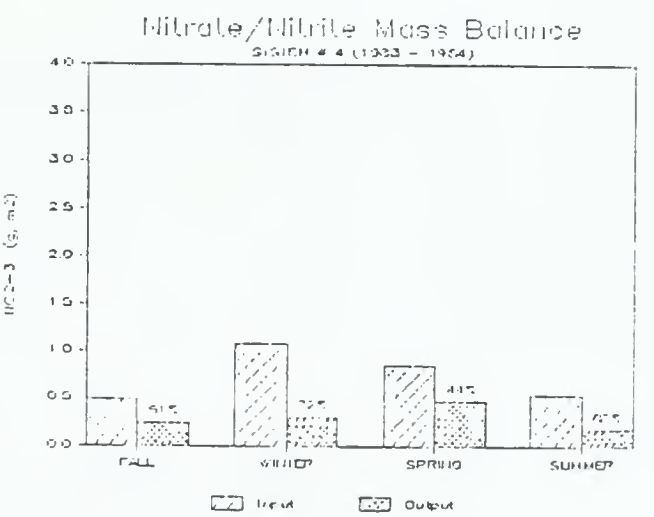

Nitrate/Nitrite Mass Balance
SYSTEM # 3 (1981 - 1982)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

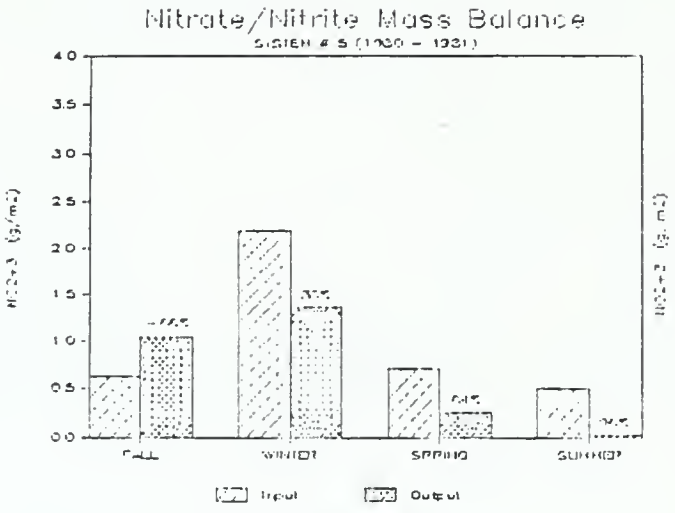

Nitrate/Nitrite Mass Balance
SYSTEM # 3 (1983 - 1984)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
Input
Output

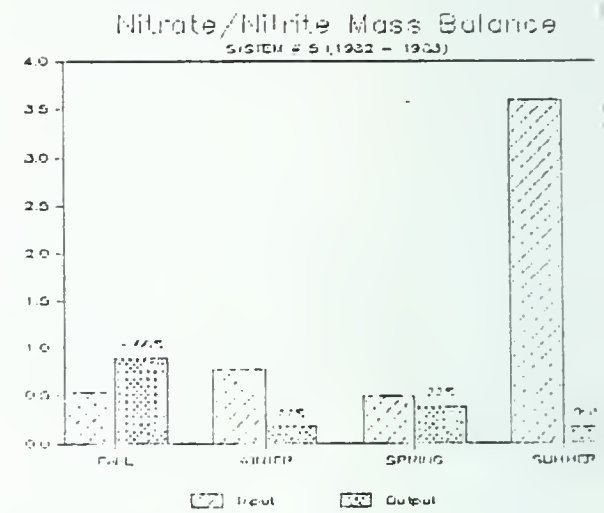

Nitrate/Nitrite Mass Balance
SYSTEM # 4 (1980 - 1981)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

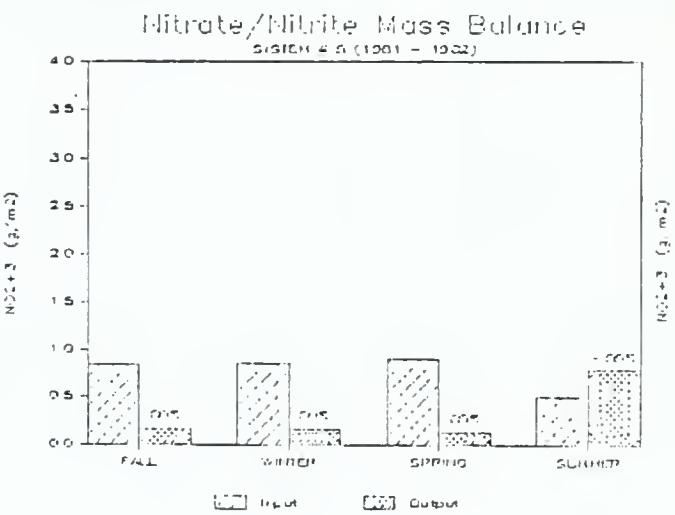

Nitrate/Nitrite Mass Balance
SYSTEM # 4 (1982 - 1983)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

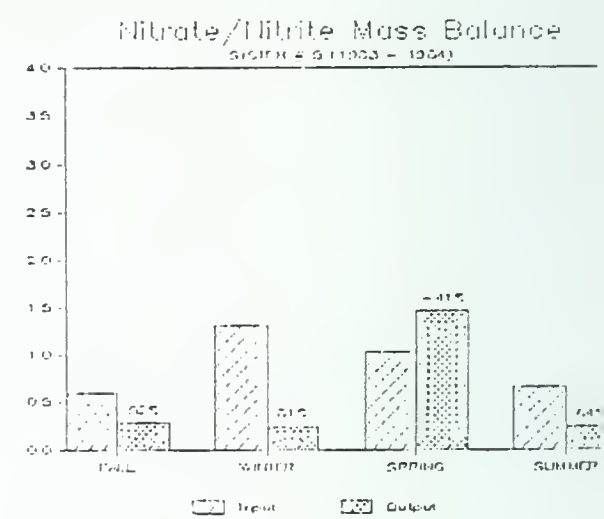

Nitrate/Nitrite Mass Balance
SYSTEM # 4 (1981 - 1982)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output

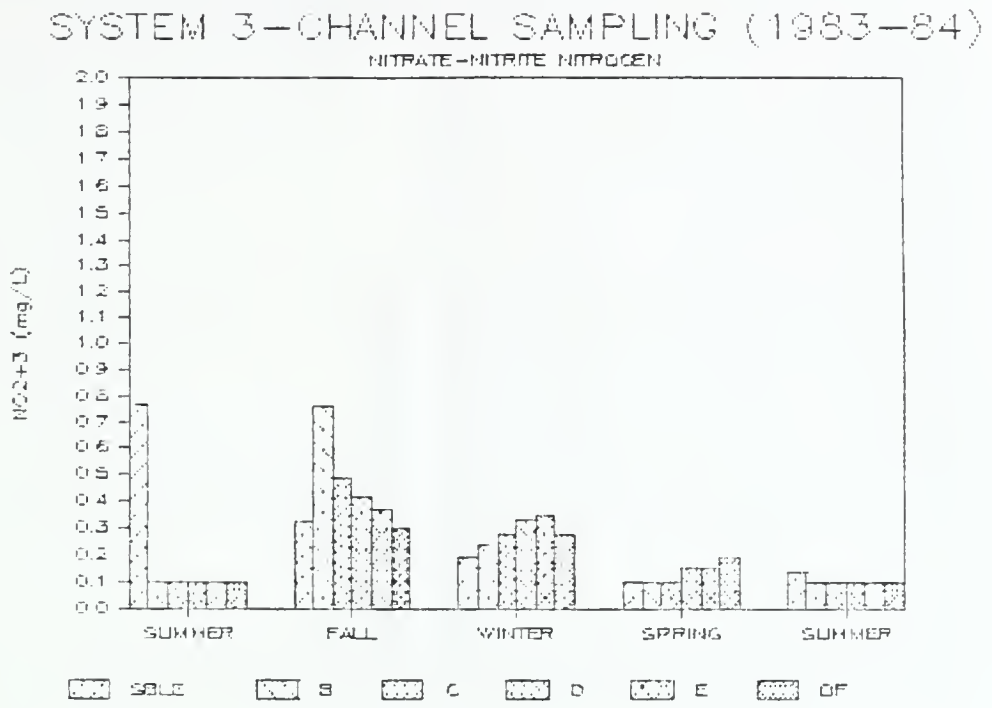

Nitrate/Nitrite Mass Balance
SYSTEM # 4 (1983 - 1984)
NO2+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output


Nitrate/Nitrite Mass Balance
SISTEN # 5 (1980 - 1981)
NO3+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output


Nitrate/Nitrite Mass Balance
SISTEM # 5 (1982 - 1983)
NO3+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output


Nitrate/Nitrite Mass Balance
SISTEM # 5 (1981 - 1982)
NO3+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output


Nitrate/Nitrite Mass Balance
SISTEM # 5 (1983 - 1984)
NO3+3 (g/m2)
4.0
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
FALL
WINTER
SPRING
SUMMER
Input
Output



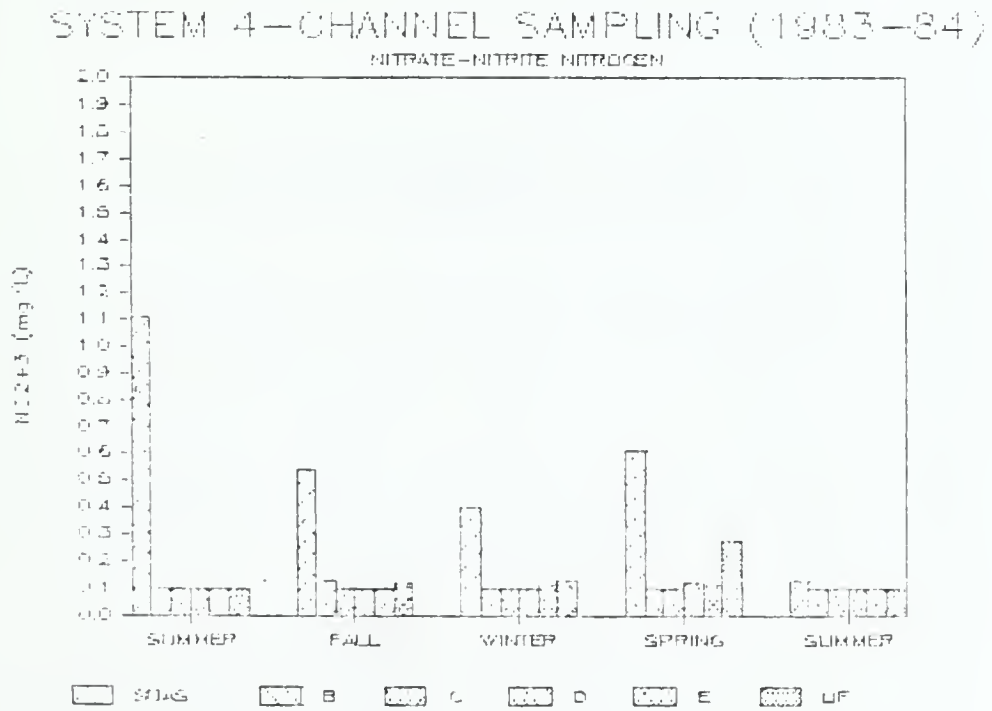
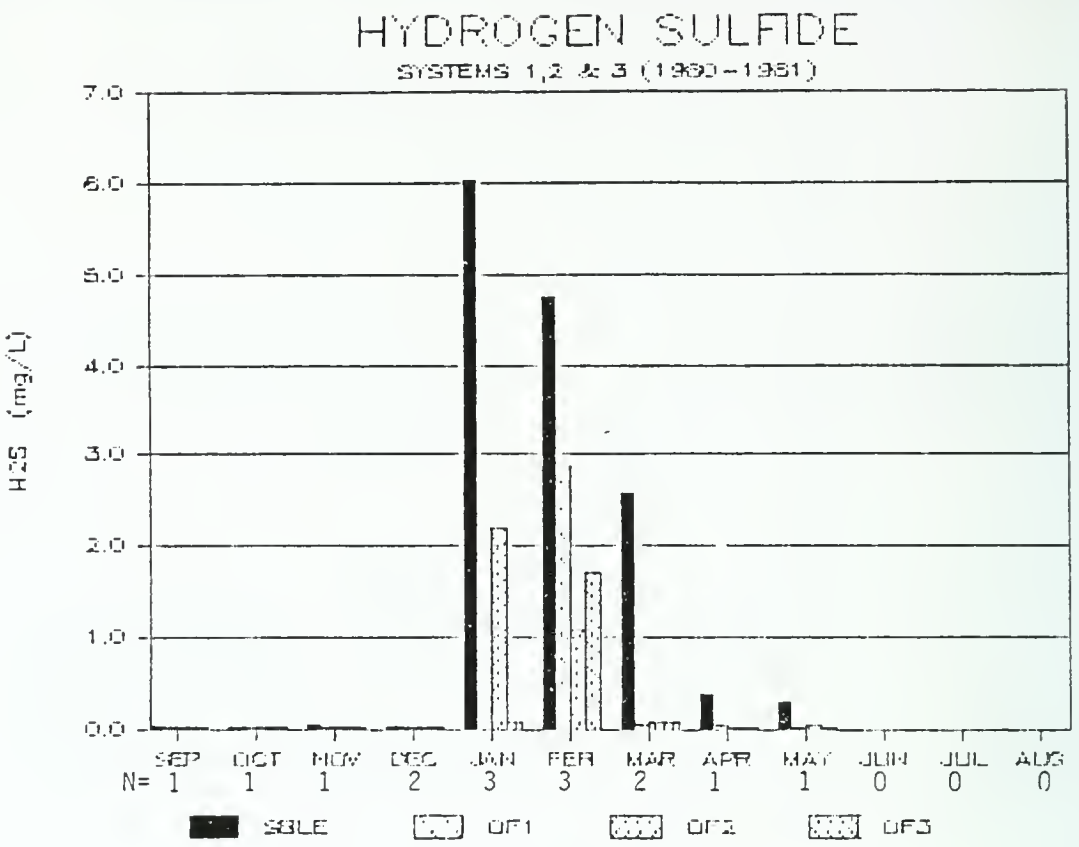


Figure A.23 Seasonal nitrate plus nitrite concentrations in the influent (SBLE or SBAS), effluent (OF) and channels (B through E) of Systems 3 and 4 (1983-4).



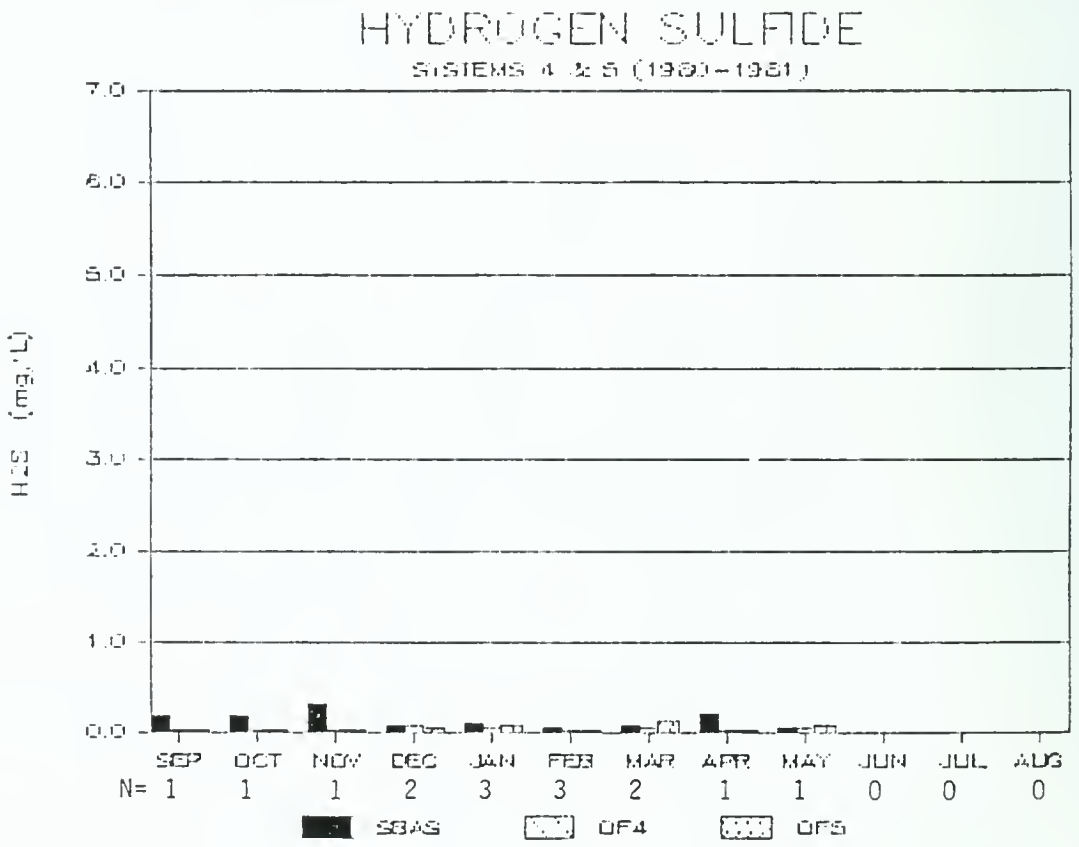
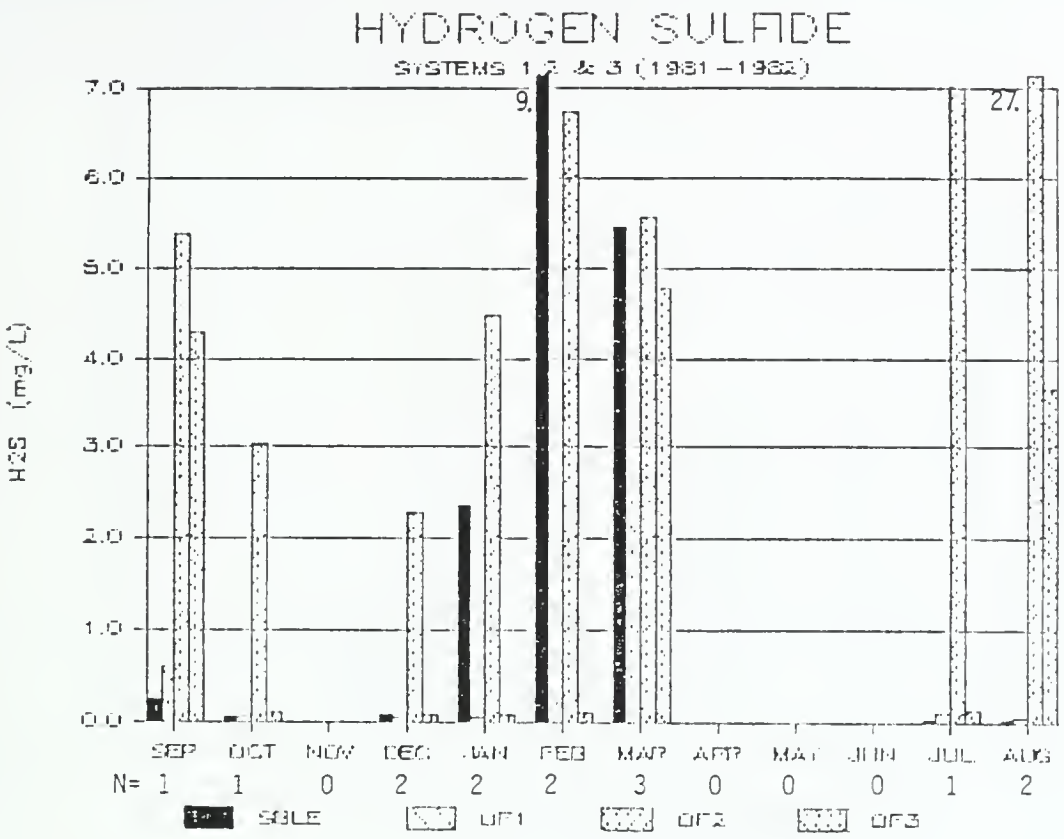
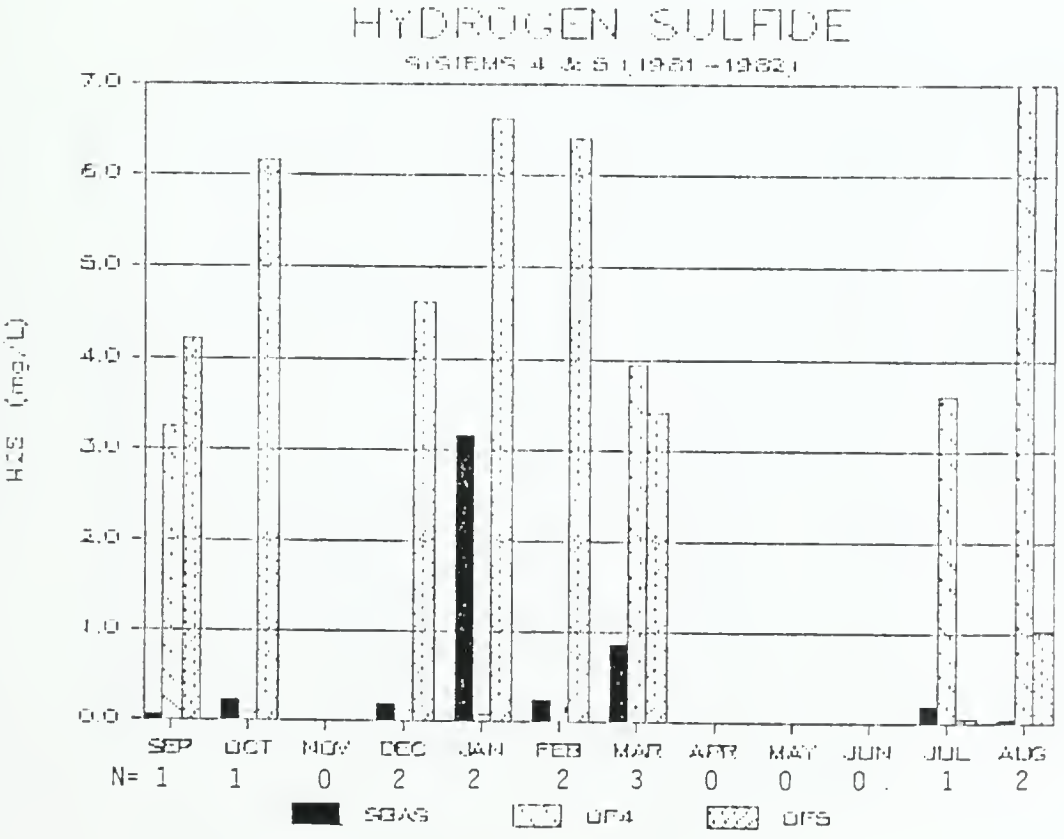
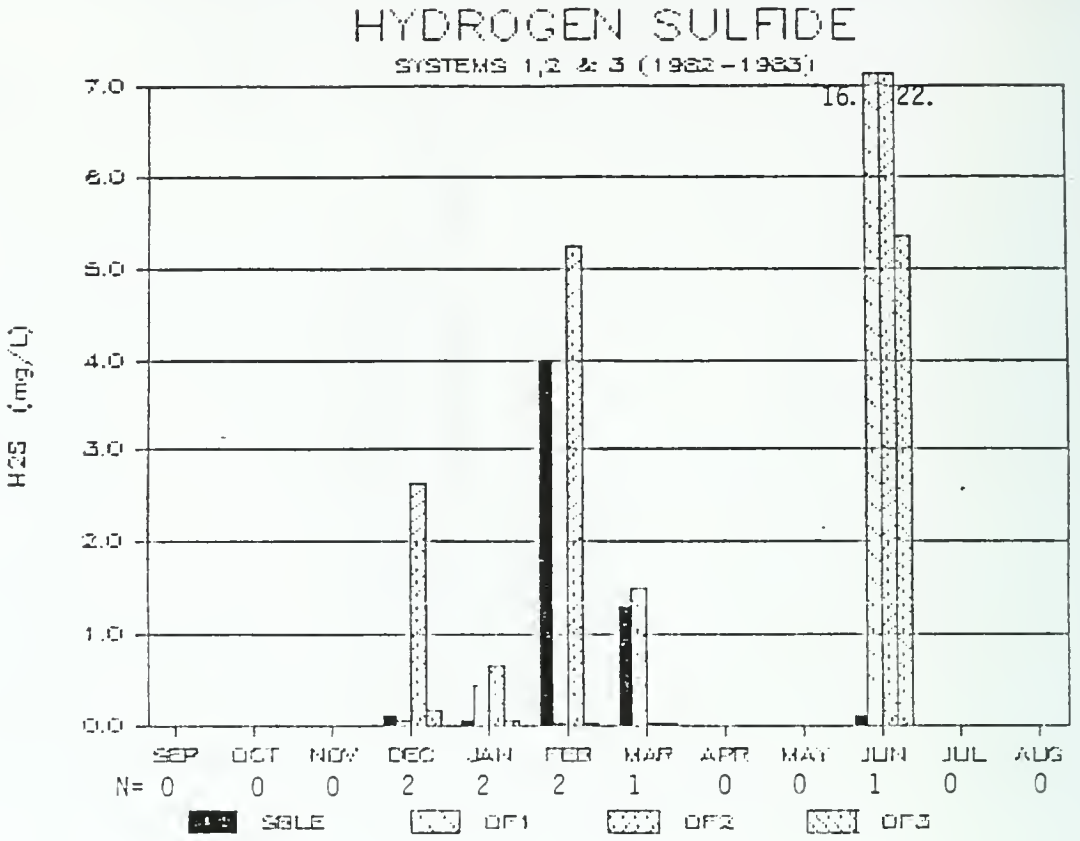
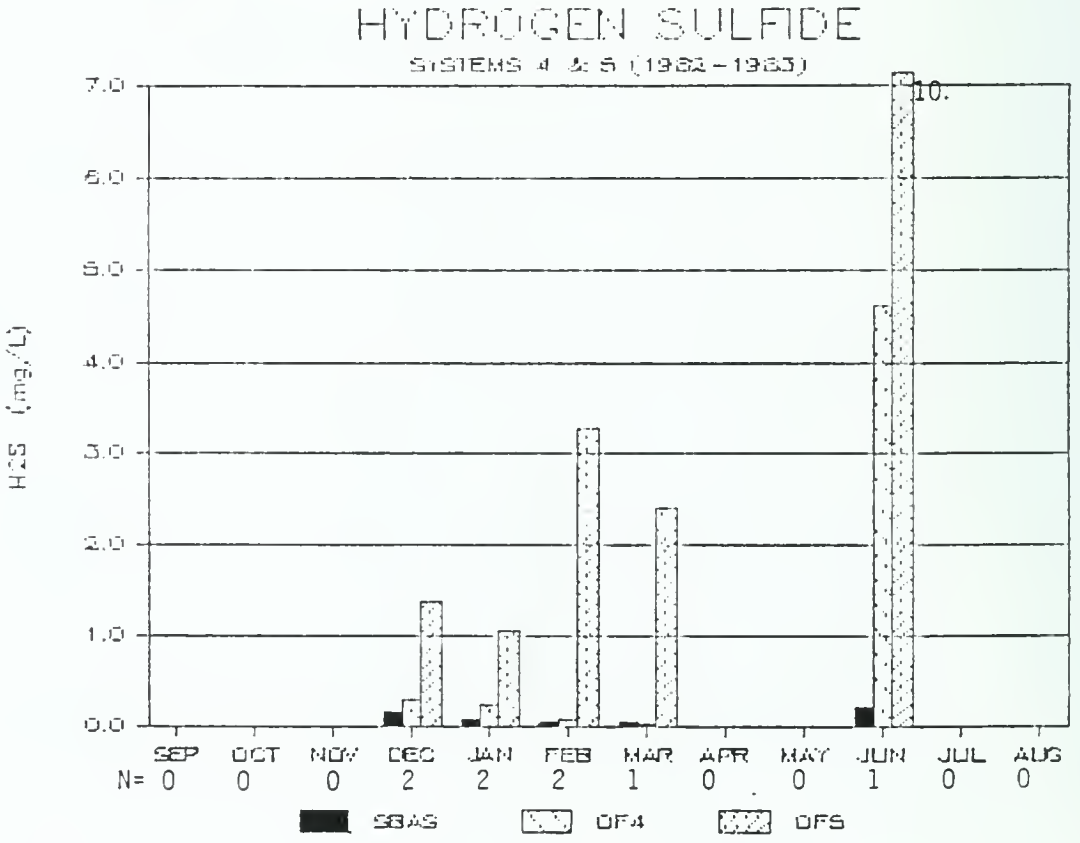


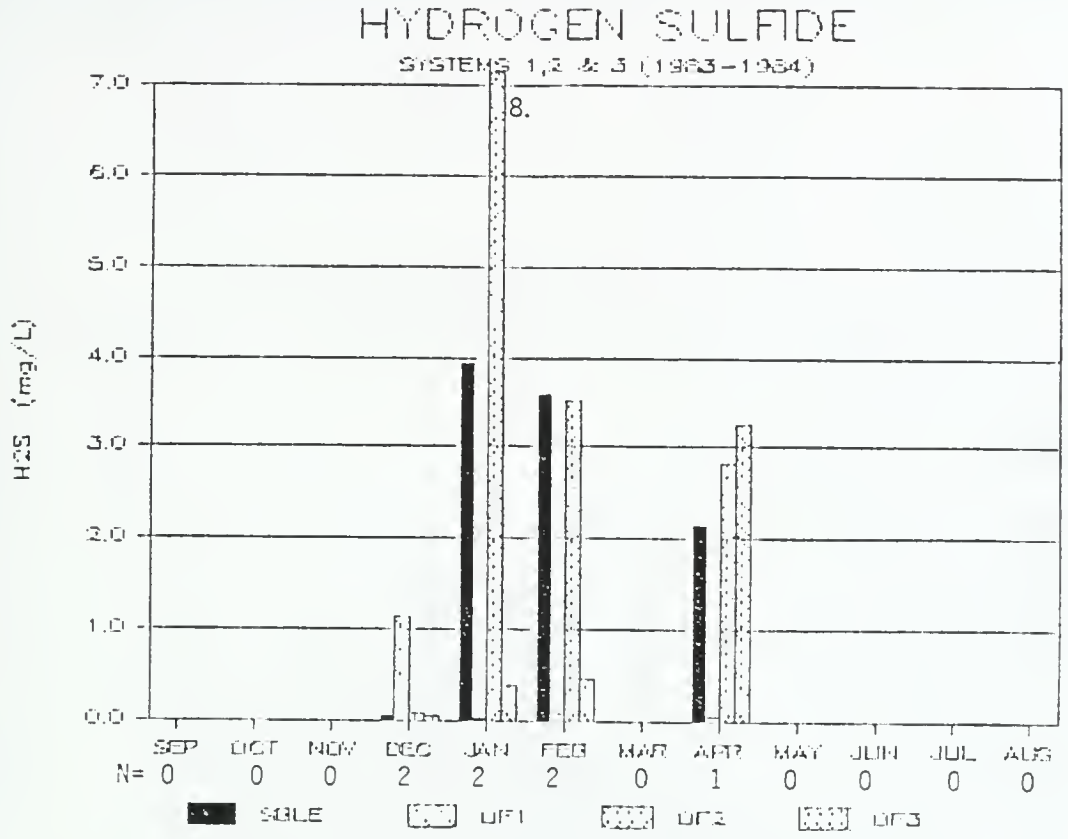
Figure A.24 Monthly average hydrogen sulfide concentrations in Systems 1-5 (1980-84). (N = no. of samples analyzed in month, SBLE = lagoon effluent, SBAS = aeration cell effluent, OF1-5 = effluent from marsh systems 1-5).

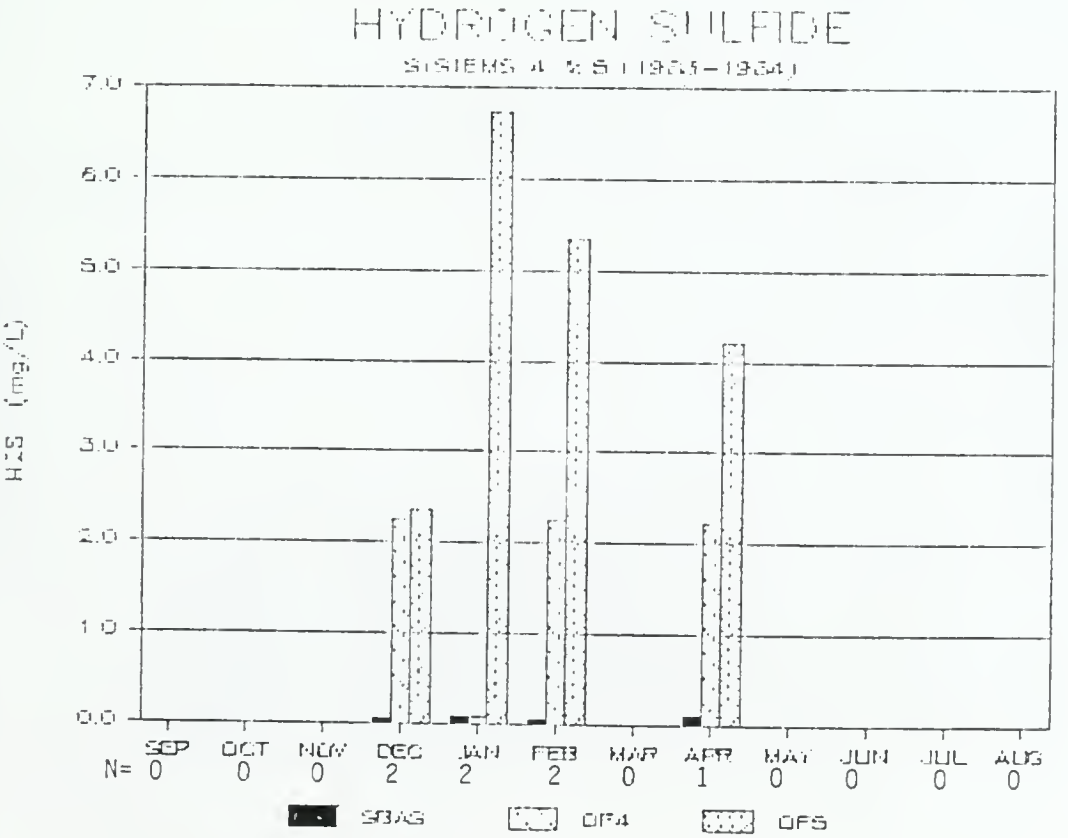

HYDROGEN SULFIDE
SYSTEMS 1, 2 & 3 (1981 – 1982)
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
9.
27.
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 0 2 2 2 3 0 0 0 1 2
SBLE
DF1
DF2
DF3


HYDROGEN SULFIDE
SYSTEMS 4 & 5 (1981 – 1982)
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 0 2 2 2 3 0 0 0 1 2
SBAS
DF4
DF5


HYDROGEN SULFIDE
SYSTEMS 1,2 & 3 (1982-1983)
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
16.
22.
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 0 0 0 2 2 2 1 0 0 1 0 0
SBLE
DF1
DF2
DF3


HYDROGEN SULFIDE
SYSTEMS 4 & 5 (1982-1983)
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
10.
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 0 0 0 2 2 2 1 0 0 1 0 0
SBAS
DF4
DF5


HYDROGEN SULFIDE
SYSTEMS 1,2 & 3 (1983-1984)
8.
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 0 0 0 2 2 2 0 1 0 0 0 0
SABLE
DF1
DF2
DF3


HYDROGEN SULFIDE
SYSTEMS 4 & 5 (1983-1984)
H2S (mg/L)
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 0 0 0 2 2 2 0 1 0 0 0 0
SB/AS
DF4
DF5

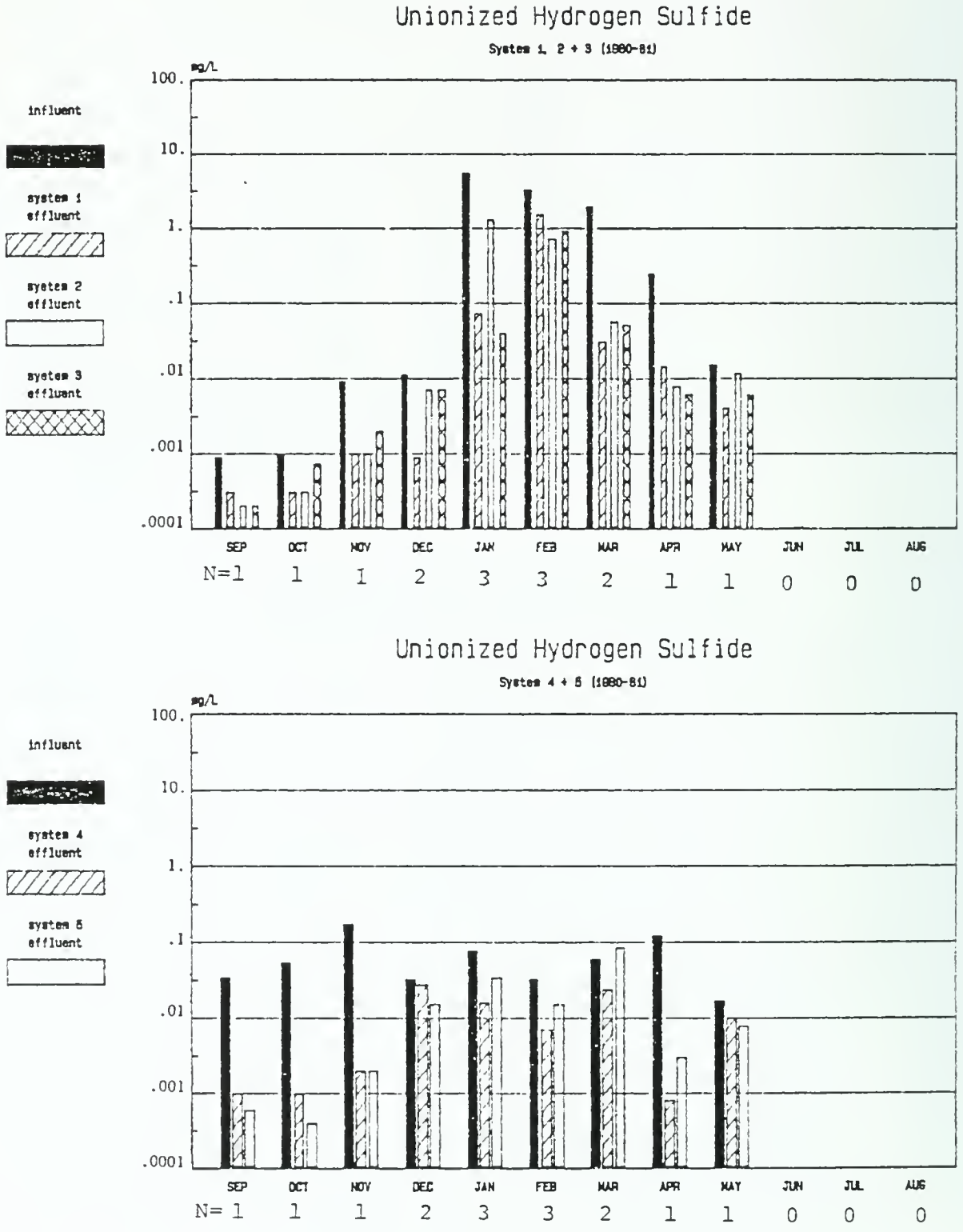
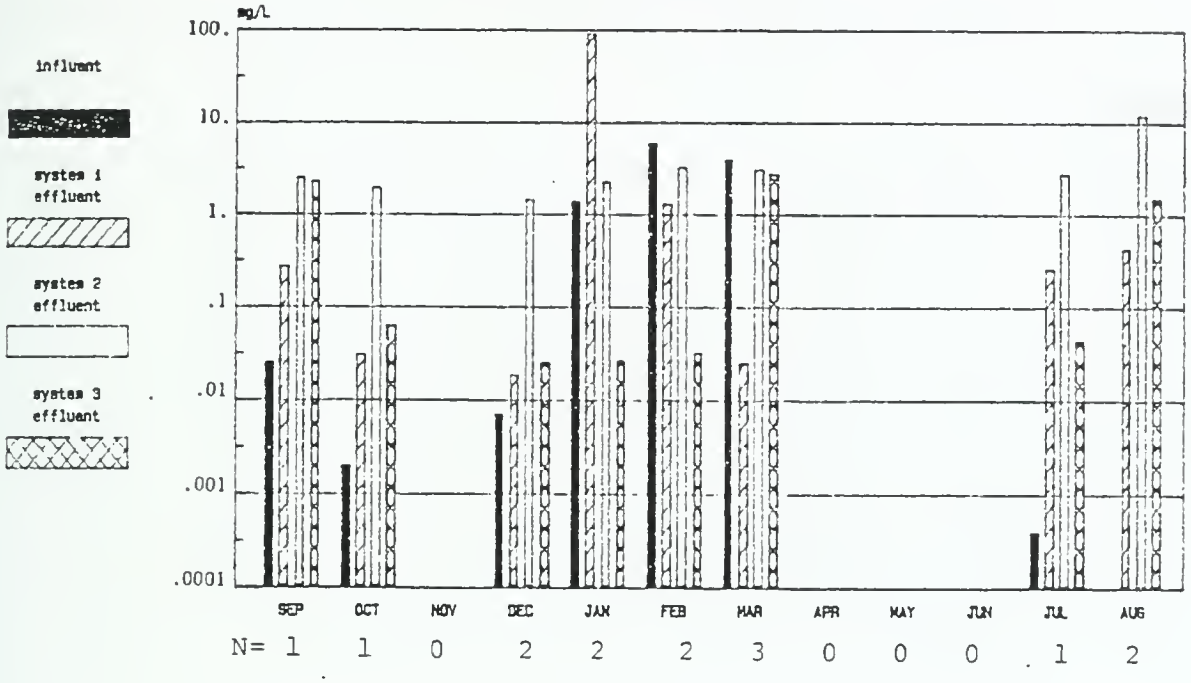


Figure A.25 Monthly average un-ionized hydrogen sulfide concentrations in Systems 1-5 (1980-84). (N = no. of samples analyzed in month, Systems 1-3 received lagoon effluent, systems 4-5 received aeration cell effluent)

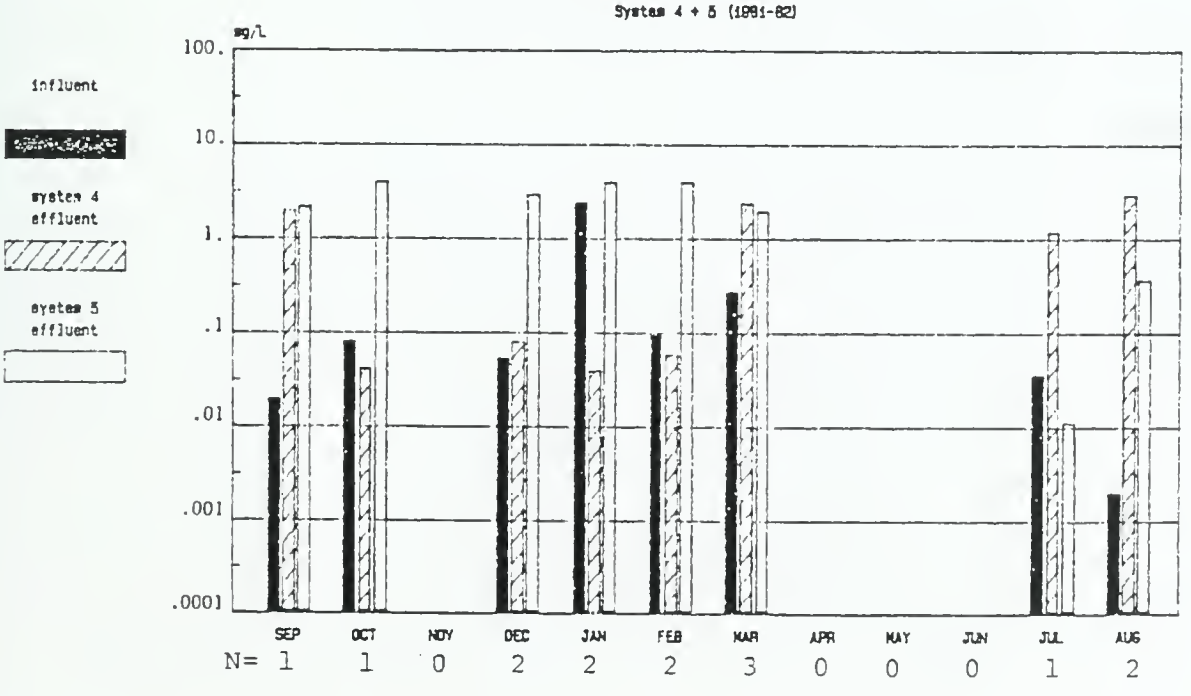
Unionized Hydrogen Sulfide

System 1, 2 + 3 (1981-82)



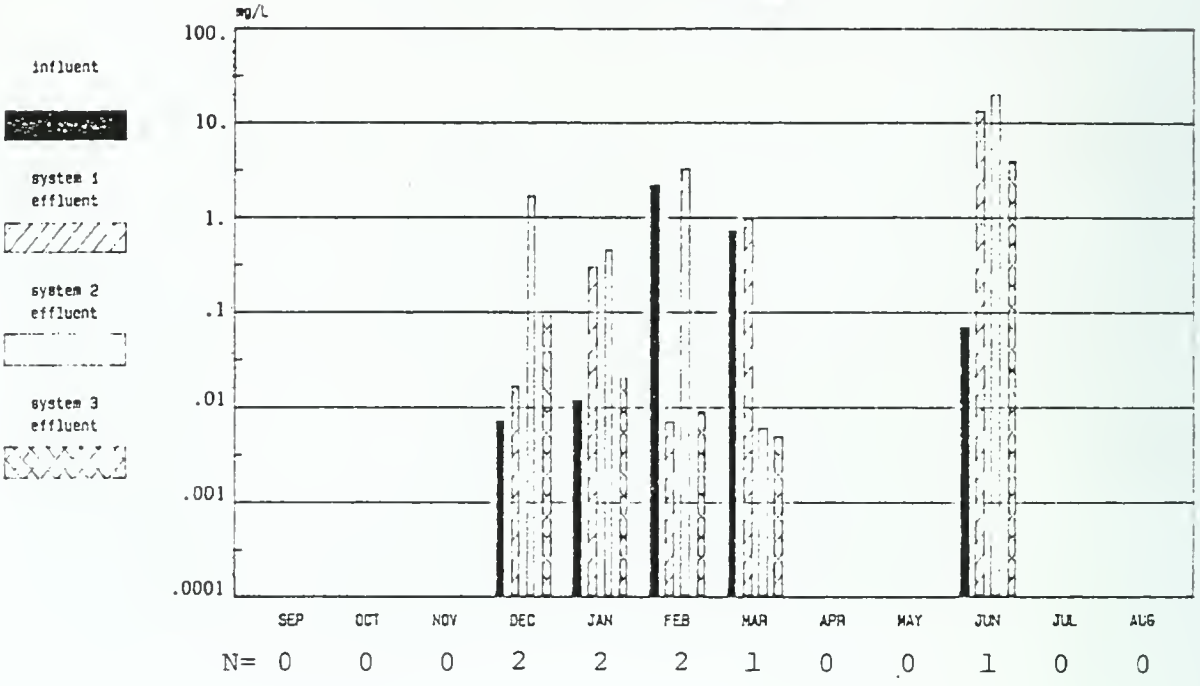
Unionized Hydrogen Sulfide

System 4 + 5 (1981-82)



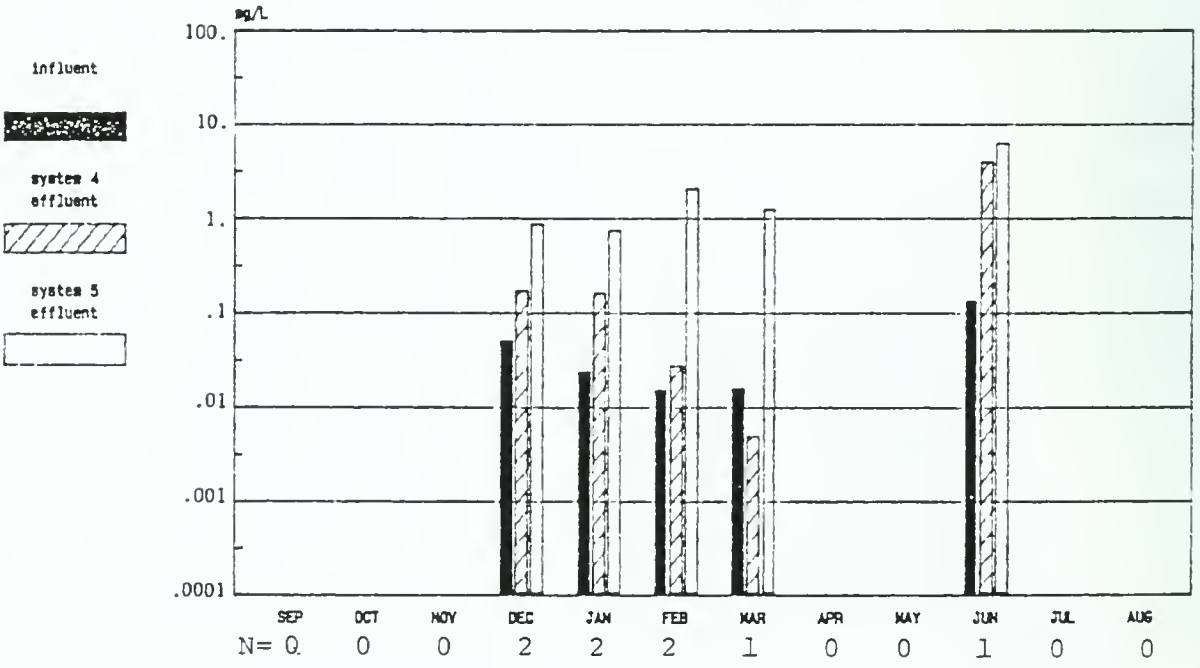
Unionized Hydrogen Sulfide

System 1, 2 + 3 (1982-83)



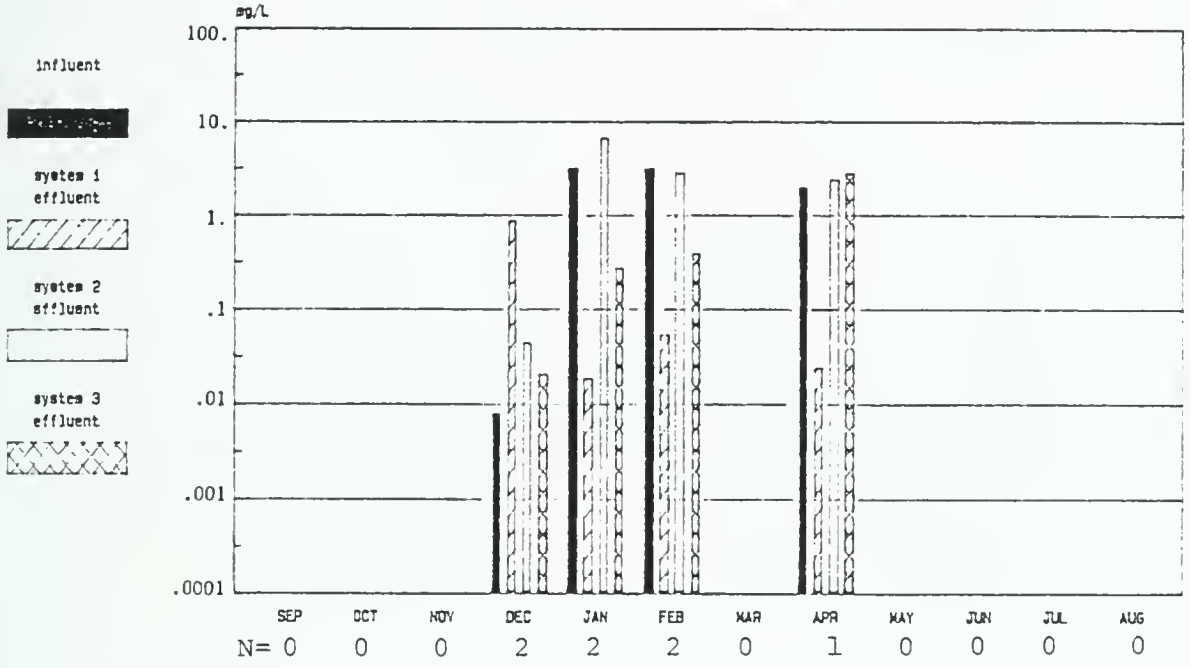
Unionized Hydrogen Sulfide

System 4 + 5 (1982-83)



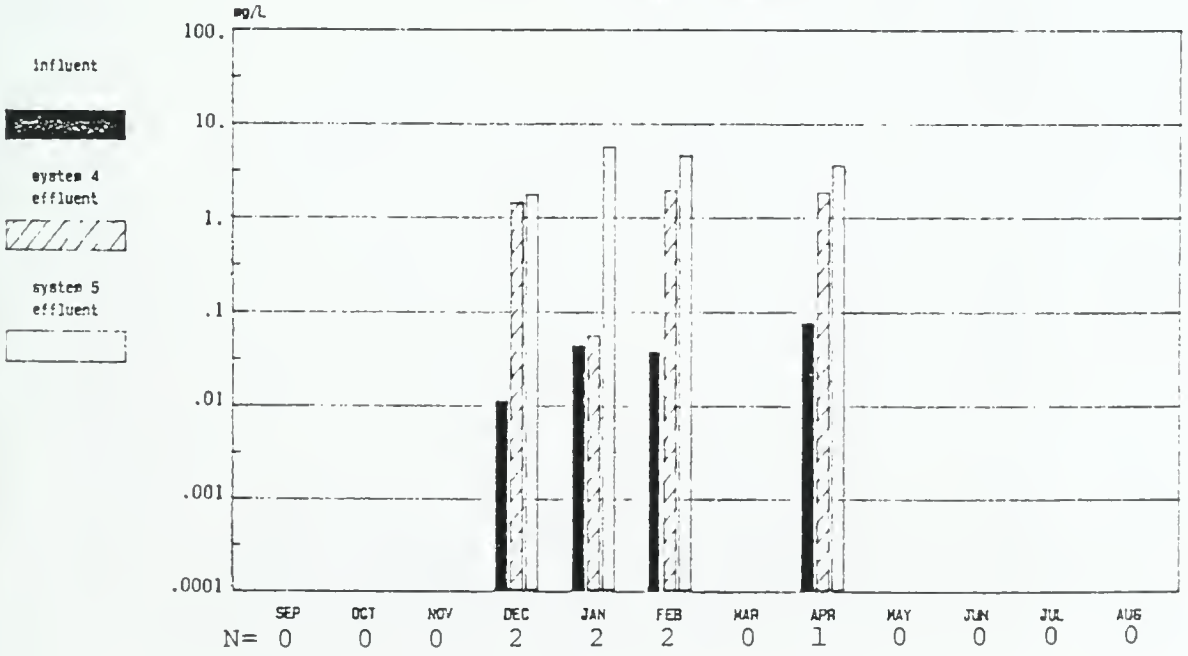
Unionized Hydrogen Sulfide

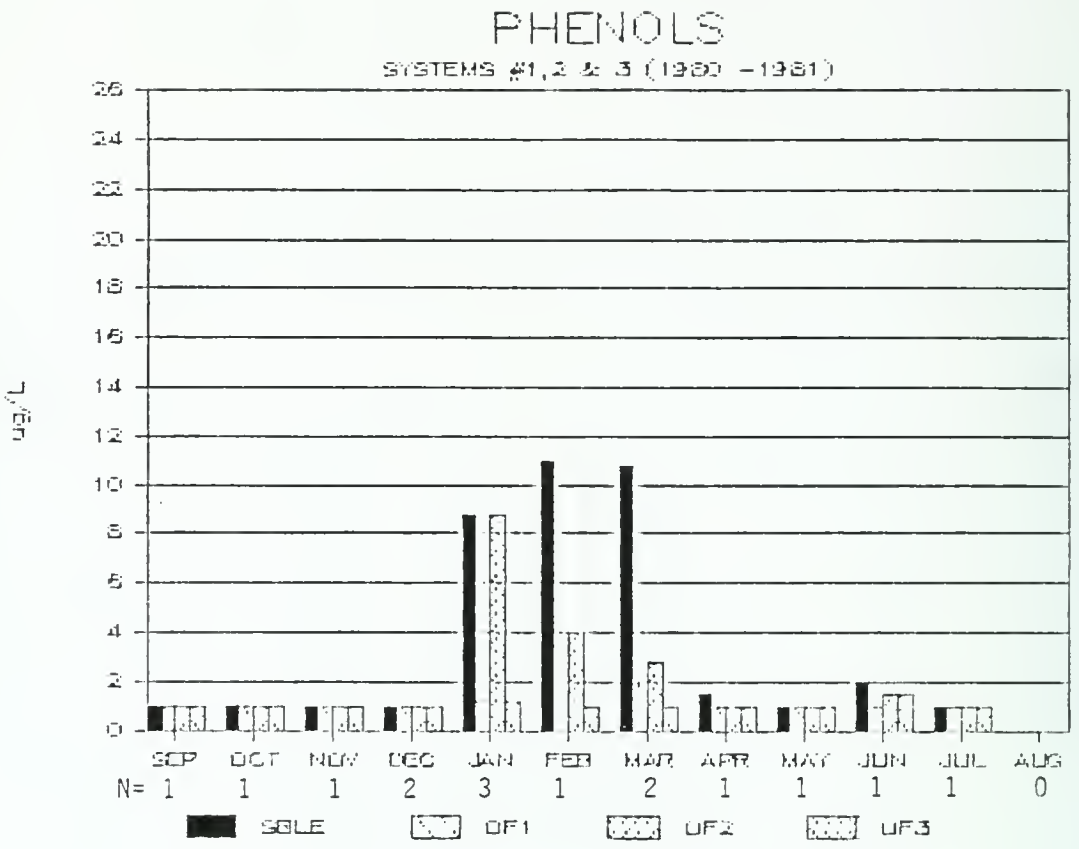
System 1, 2 + 3 (1983-84)



Unionized Hydrogen Sulfide

System 4 + 5 (1983-84)





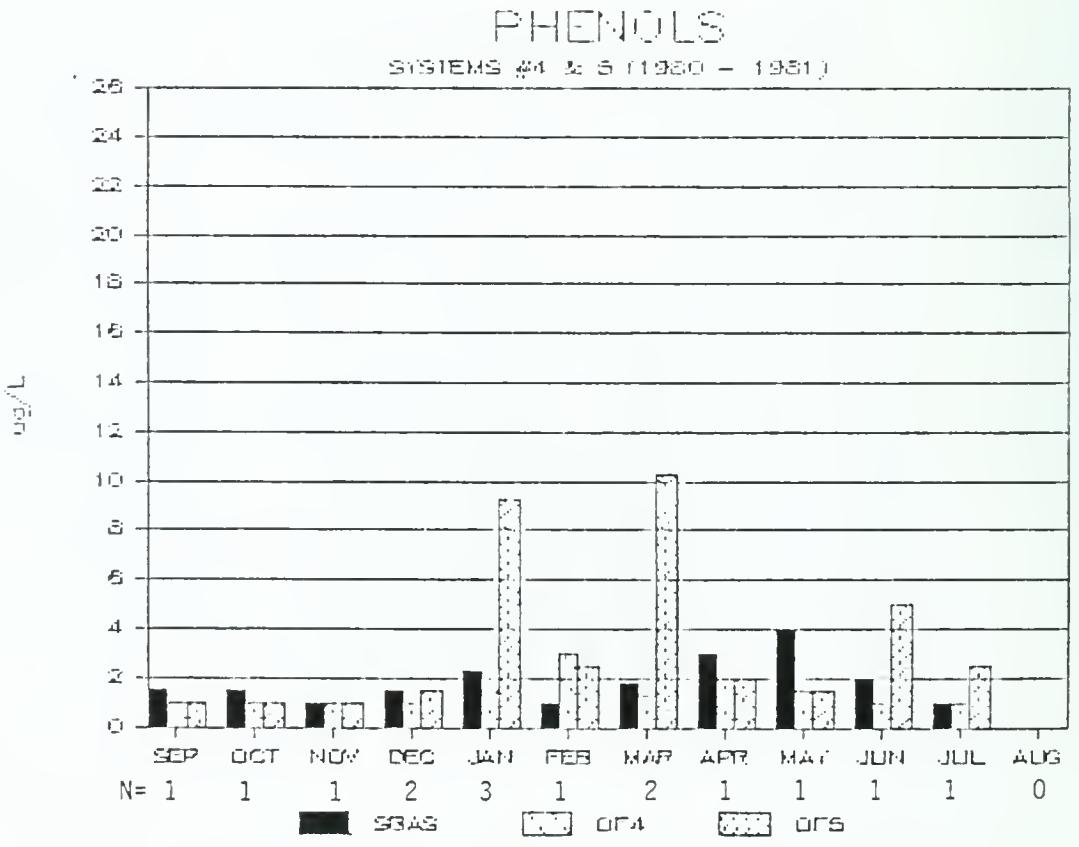
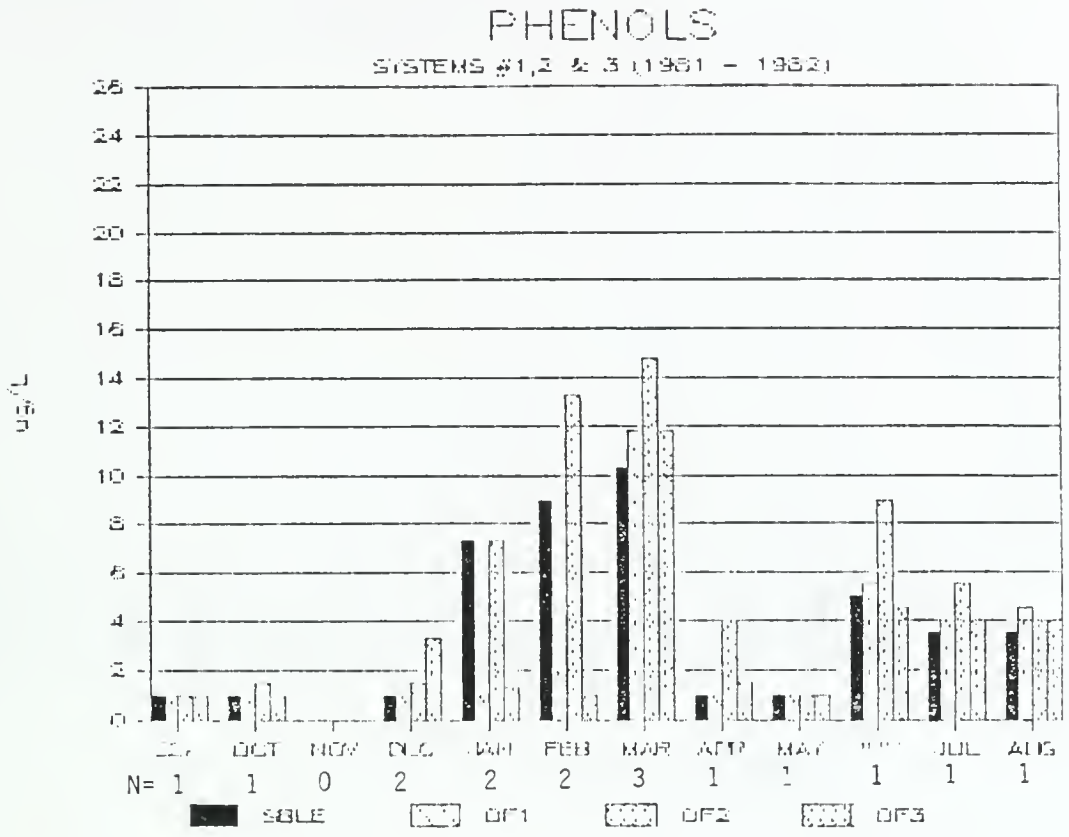
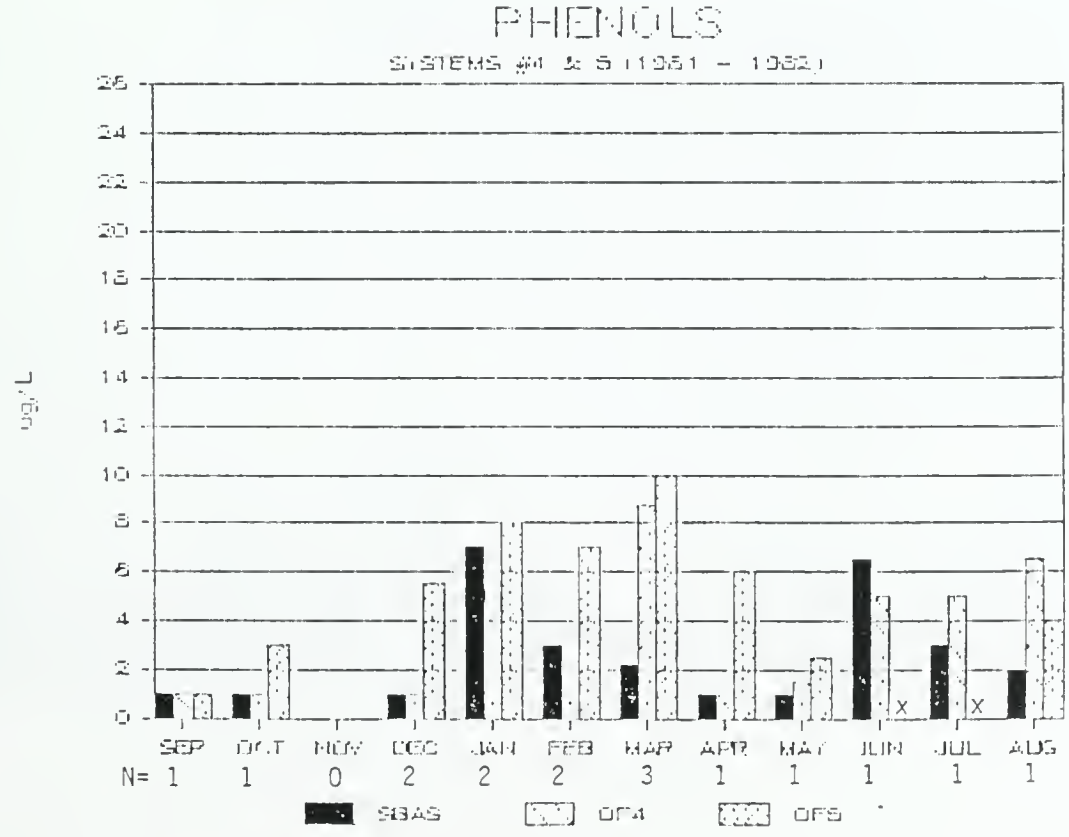
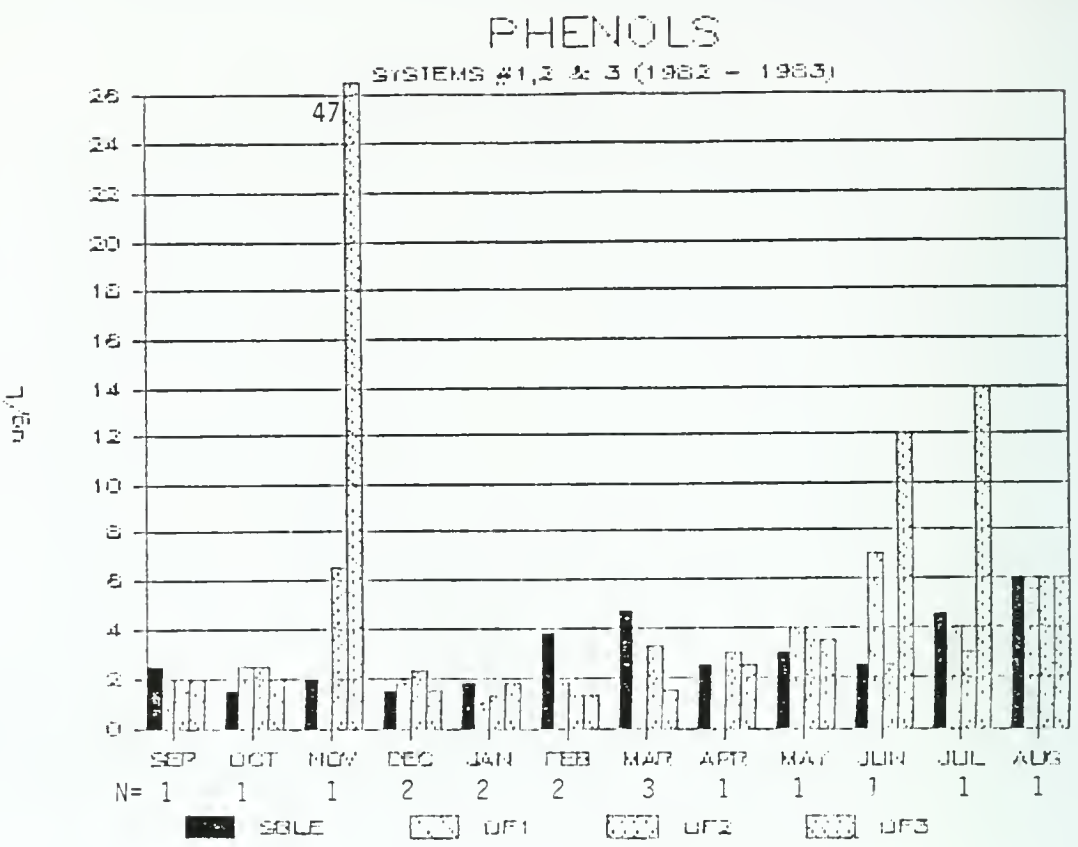
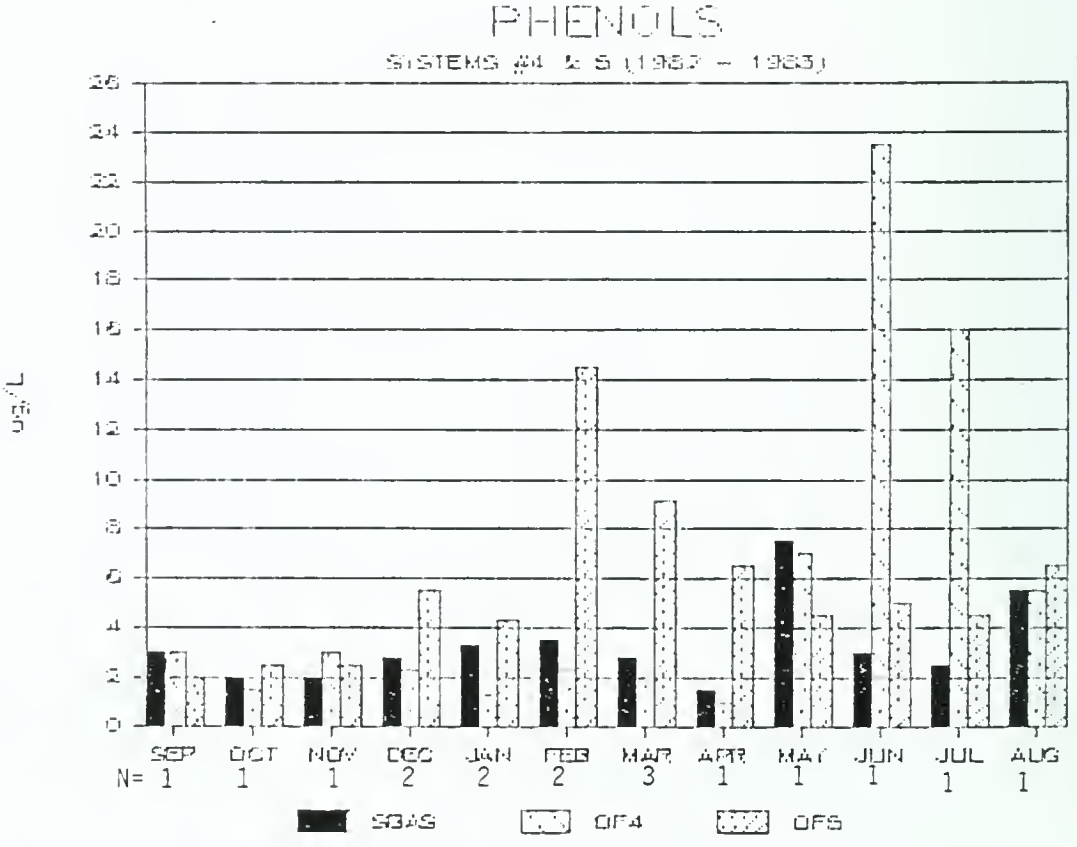


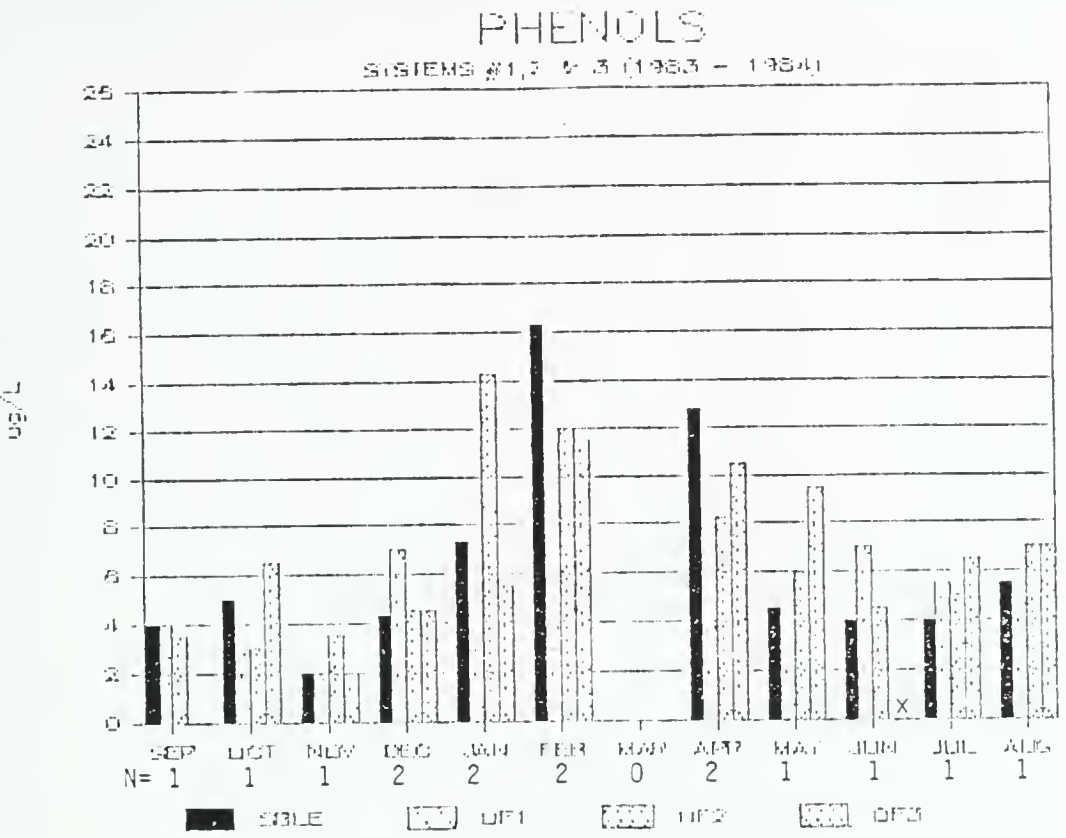
Figure A.26 Monthly average phenol concentrations in Systems 1-5 (1980-84).
(N = no. of samples analyzed in month,
SBLE = lagoon effluent, SBAS = aeration cell effluent,
x = data not available) - 226

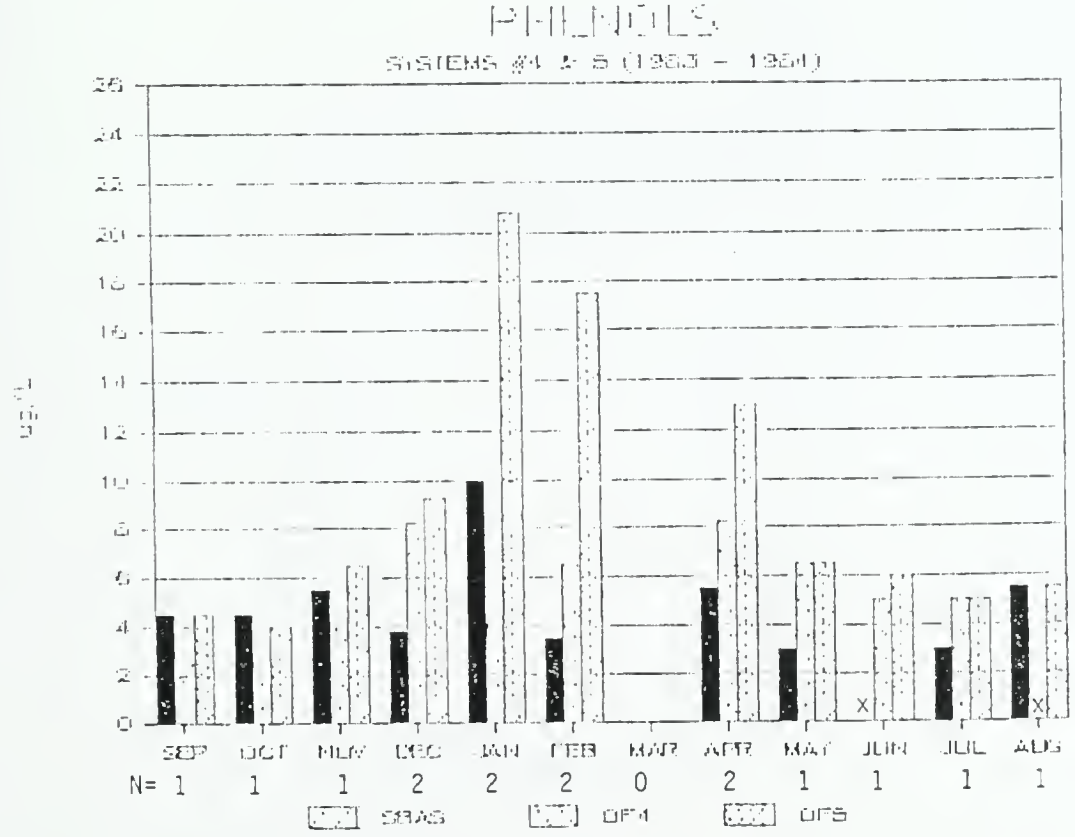

PHENOLS
SYSTEMS #1,2 & 3 (1981 - 1982)
ug/L
26
24
22
20
18
16
14
12
10
8
6
4
2
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 0 2 2 2 3 1 1 1 1 1
SBLE
OF1
OF2
OF3


PHENOLS
SYSTEMS #4 & 5 (1981 - 1982)
ug/L
26
24
22
20
18
16
14
12
10
8
6
4
2
0
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 0 2 2 2 3 1 1 1 1 1
SBAS
OF4
OF5


PHENOLS
SYSTEMS #1,2 & 3 (1982 - 1983)
ug/L
47
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 1 2 2 2 3 1 1 1 1 1
SBLE
OF1
OF2
OF3


PHENOLS
SYSTEMS #4 & 5 (1982 - 1983)
ug/L
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 1 2 2 2 3 1 1 1 1 1
SBAS
OF4
OF5


PHENOLS
SYSTEMS #1,3 & 5 (1983 - 1984)
ug/L
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 1 2 2 2 0 2 1 1 1 1
INFLUENT
DIF1
DIF2
DIF3


PHENOLS
SYSTEMS #1 & 5 (1983 - 1984)
ug/L
SEP OCT NOV DEC JAN FEB MAR APR MAY JUN JUL AUG
N= 1 1 1 2 2 2 0 2 1 1 1 1
SEWAGE
DIF1
DIF5

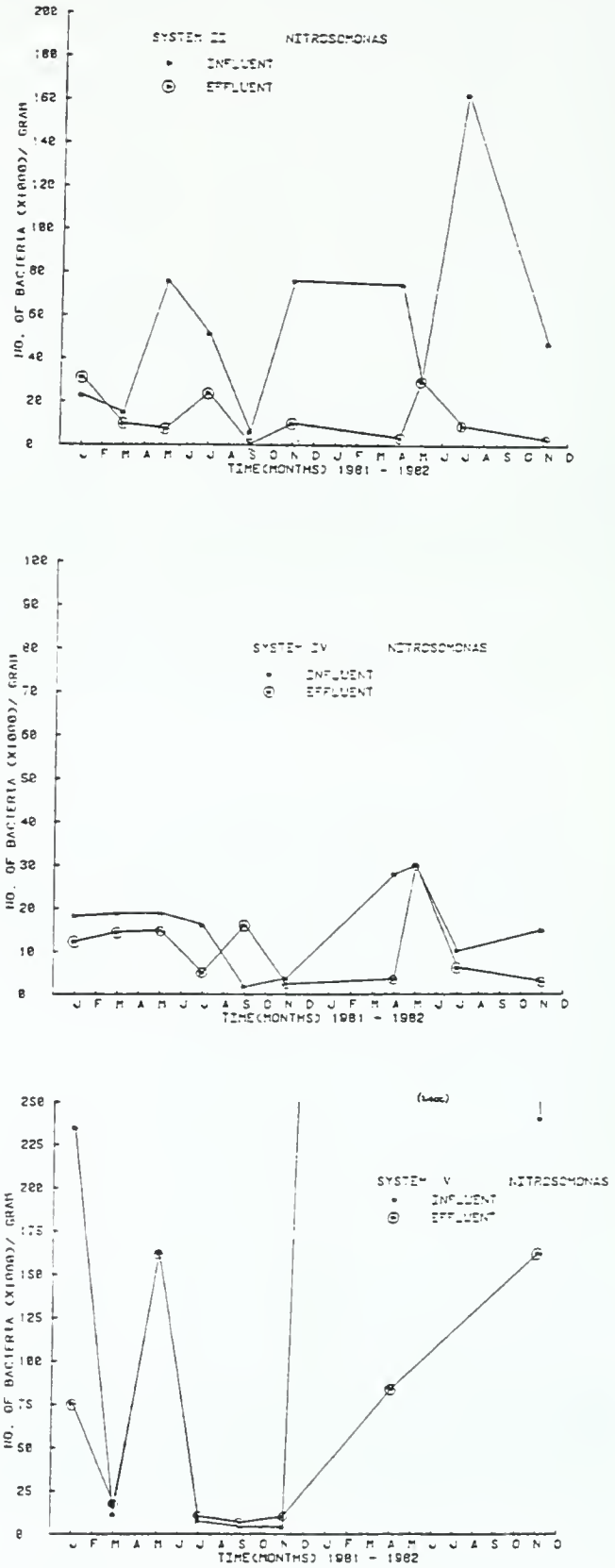


Figure A.27 Nitrosomonas numbers in the sediment of Systems 2, 4 and 5 in 1981-82.

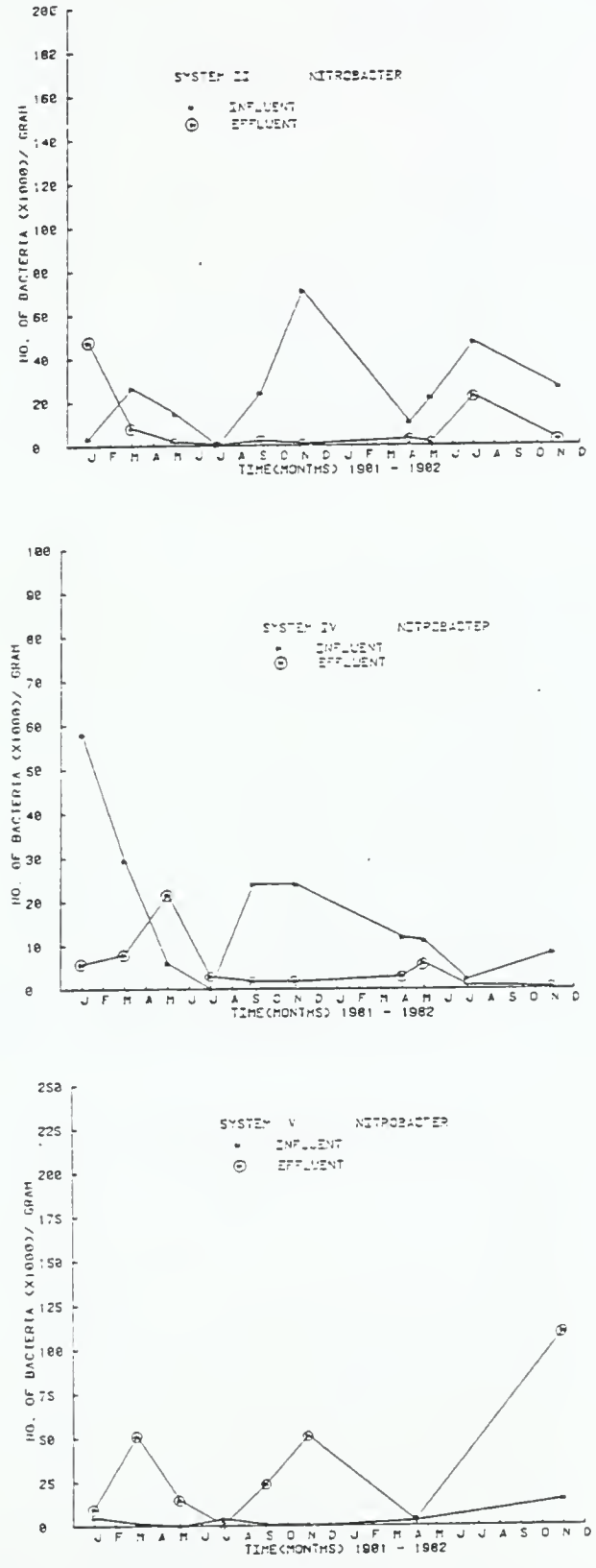


Figure A.28 Nitrobacter numbers in the sediment of Systems 2, 4 and 5 in 1981-82.

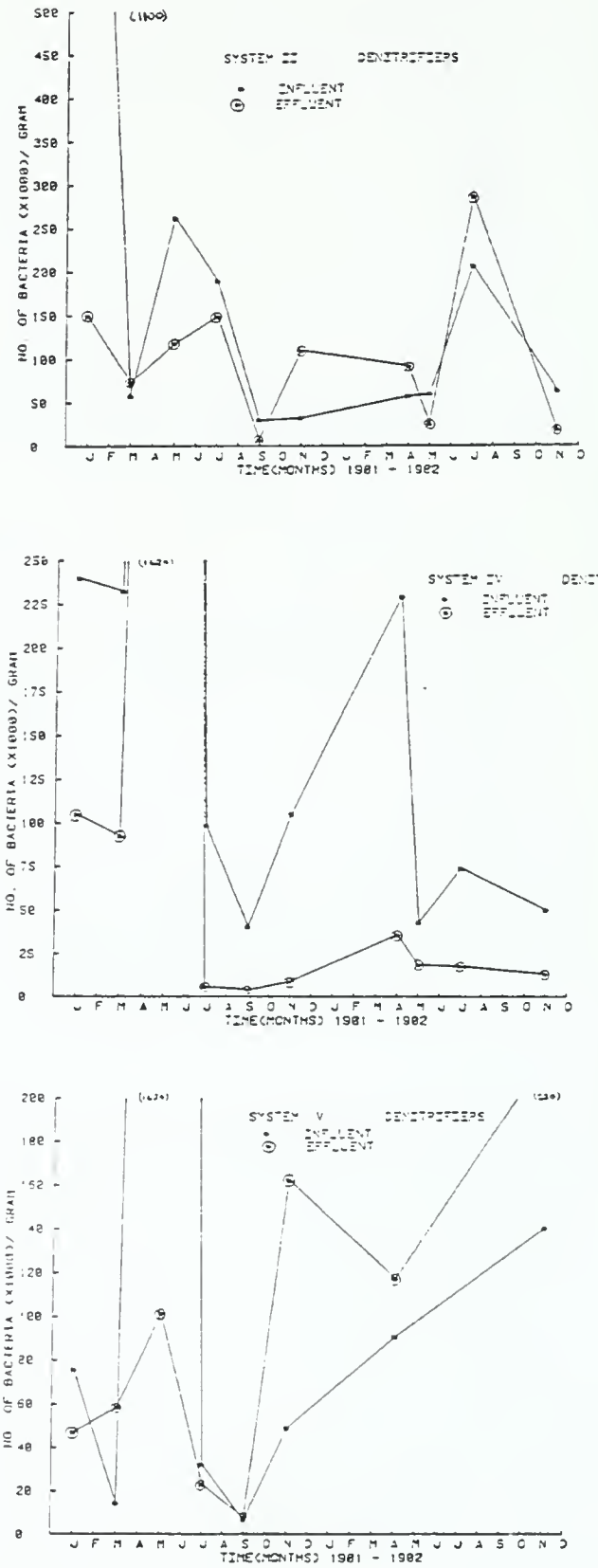


Figure A.29 Numbers of denitrifying bacteria in the sediment of Systems 2, 4 and 5 in 1981-82.

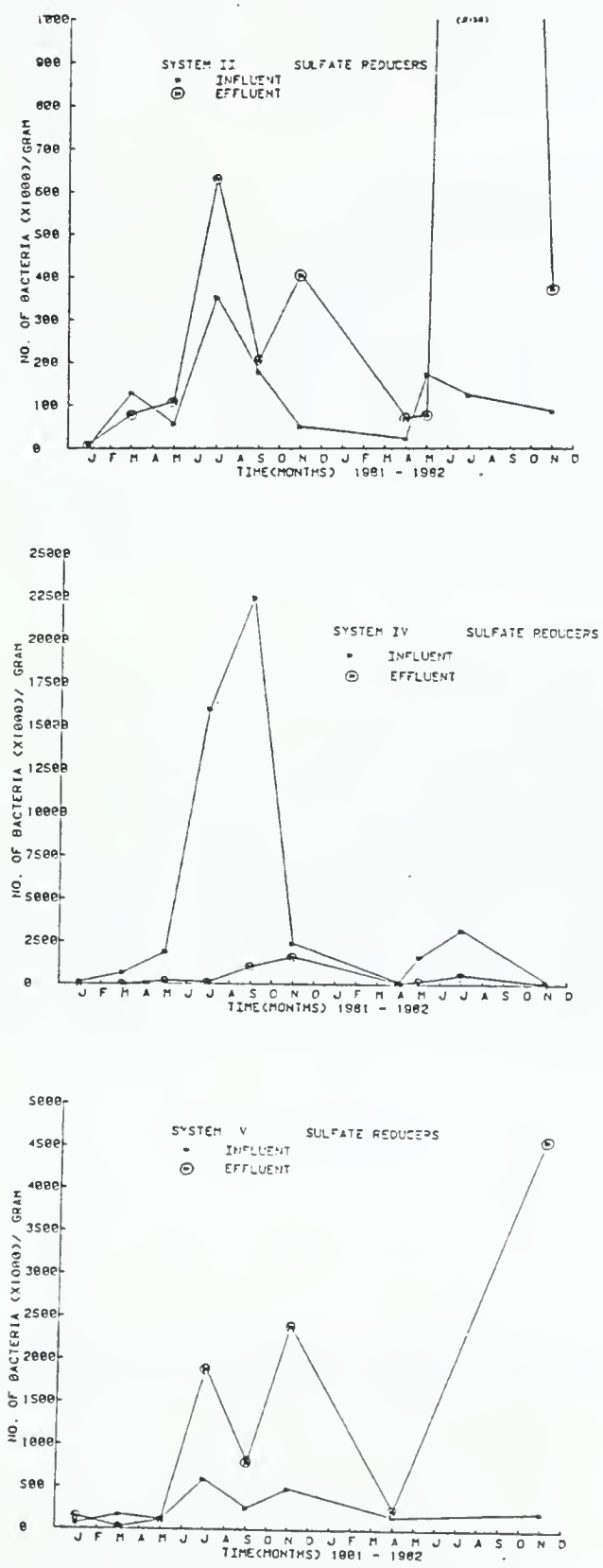


Figure A.30 Numbers of sulfate-reducing bacteria in the sediment of Systems 2, 4 and 5 in 1981-82.

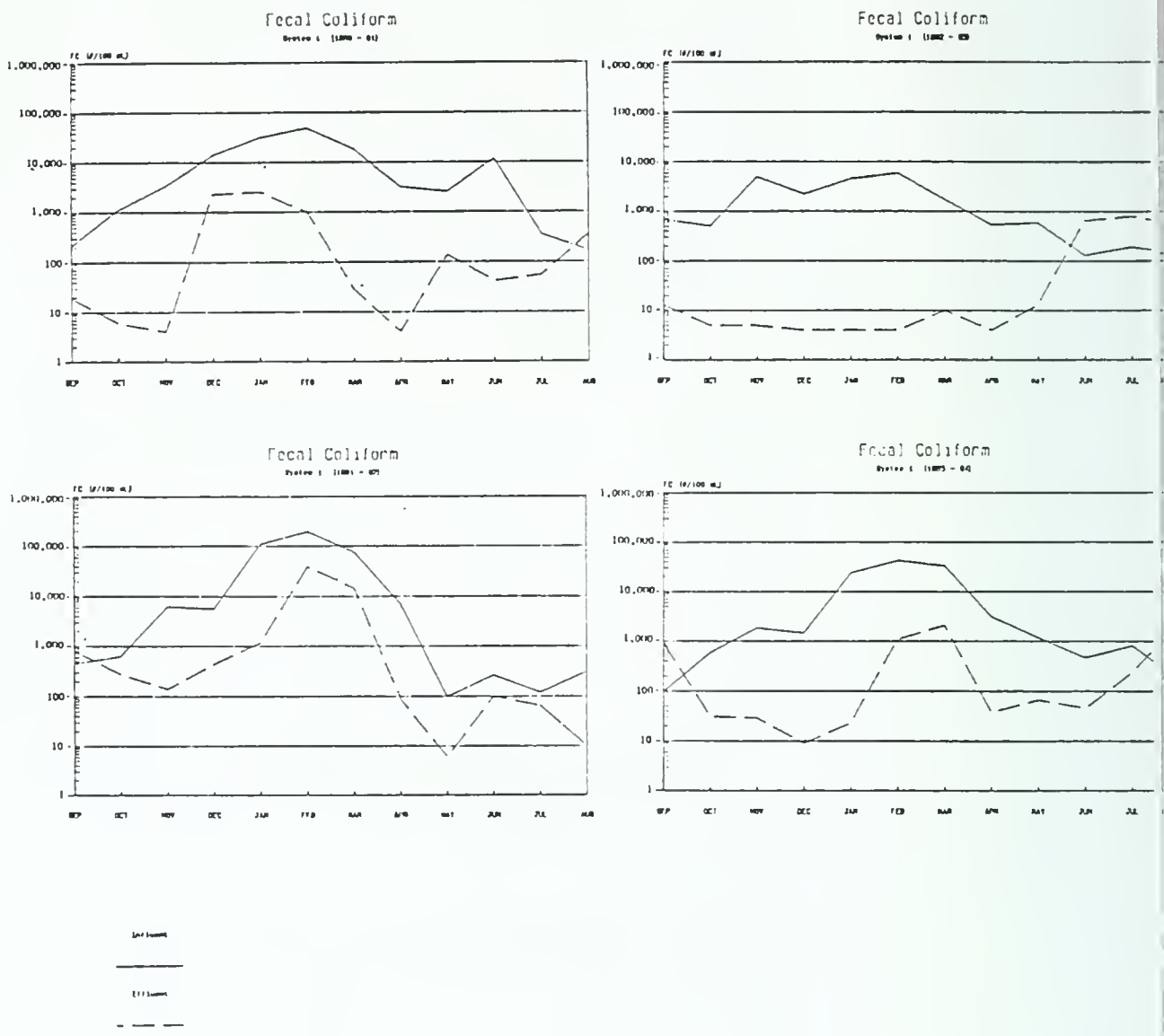
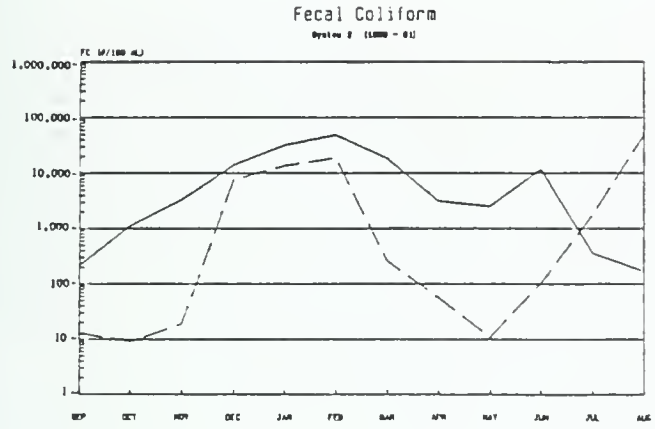
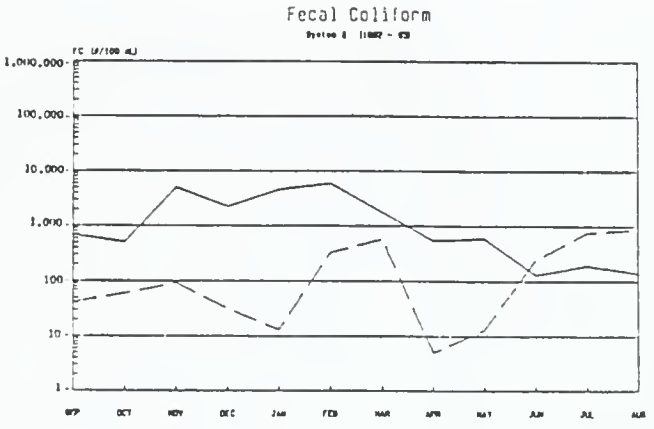
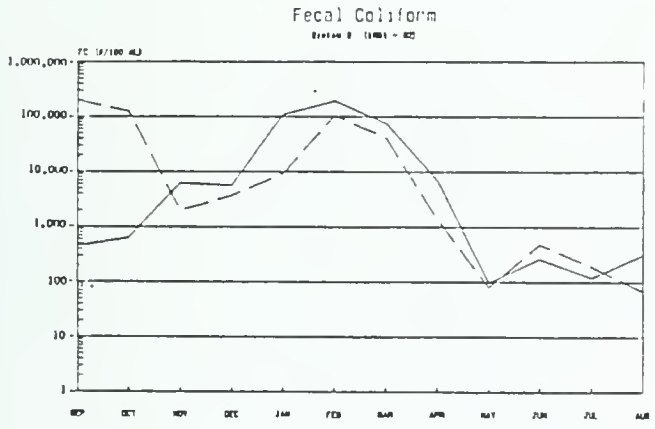
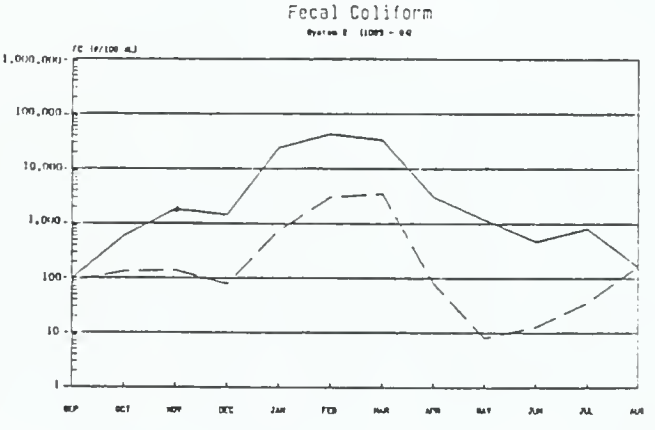


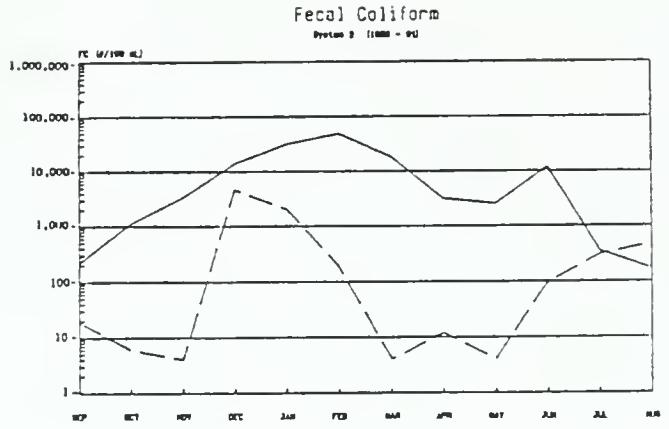
Figure A.31 Monthly geometric mean fecal coliform levels in Systems 1-5 (4 years).

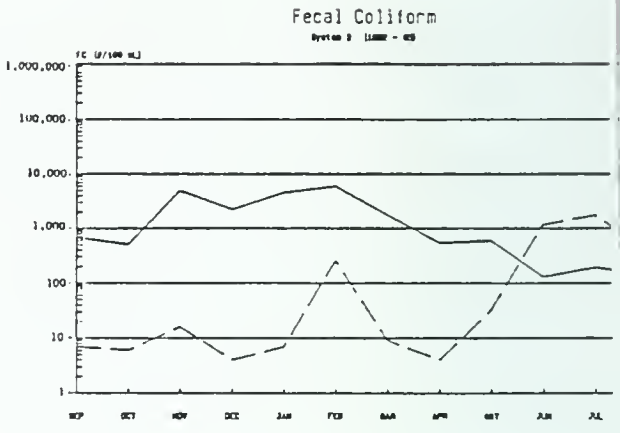

Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG

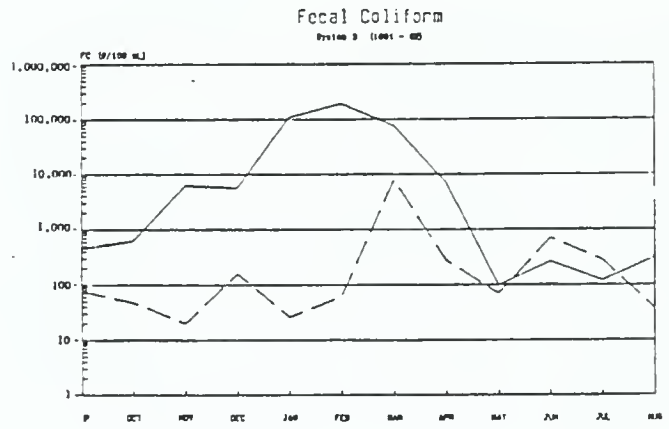

Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG

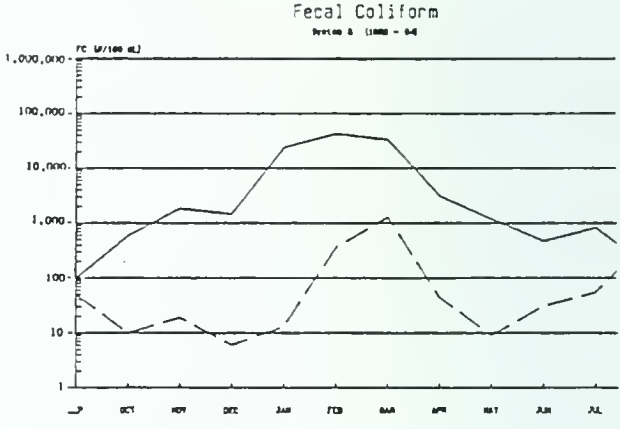

Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG


Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1
SEP
OCT
NOV
DEC
JAN
FEB
MAR
APR
MAY
JUN
JUL
AUG


Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1


Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1


Fecal Coliform
1,000,000
100,000
10,000
1,000
100
10
1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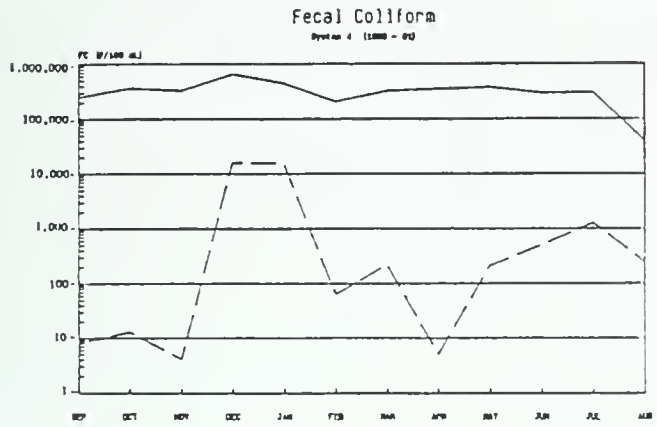
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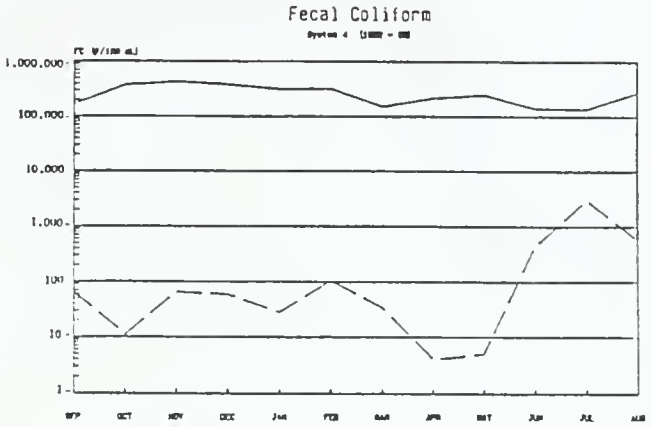
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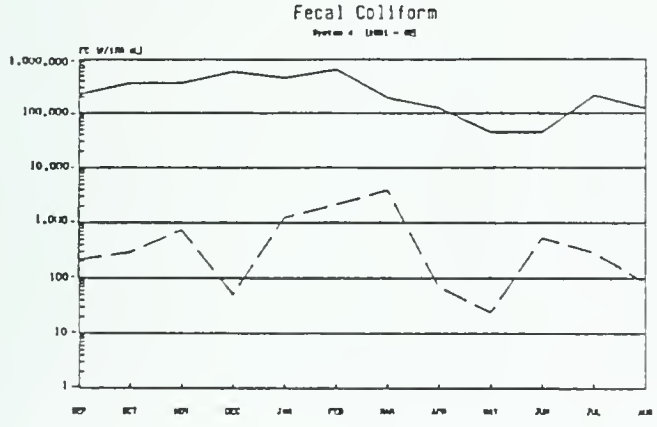
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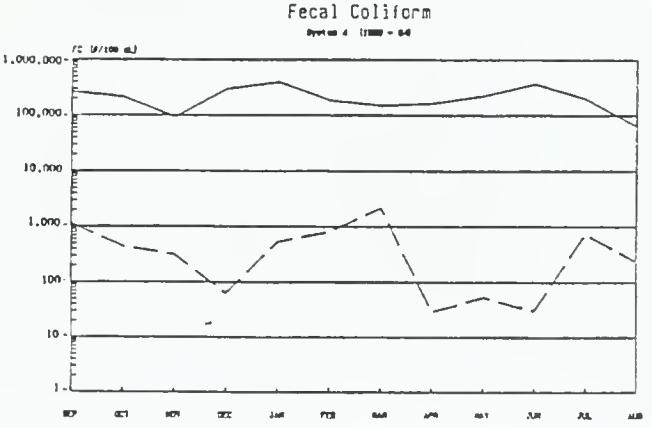
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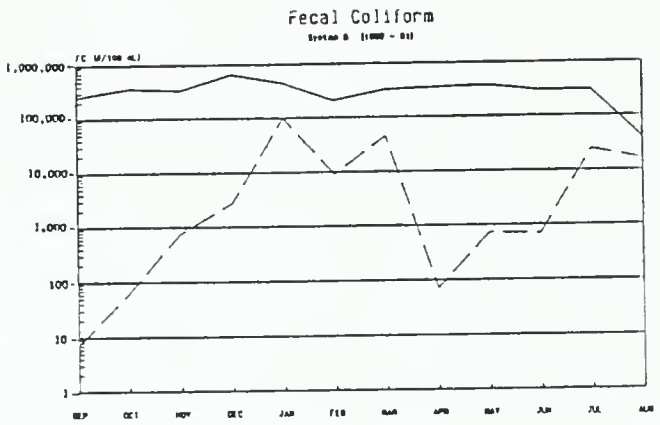
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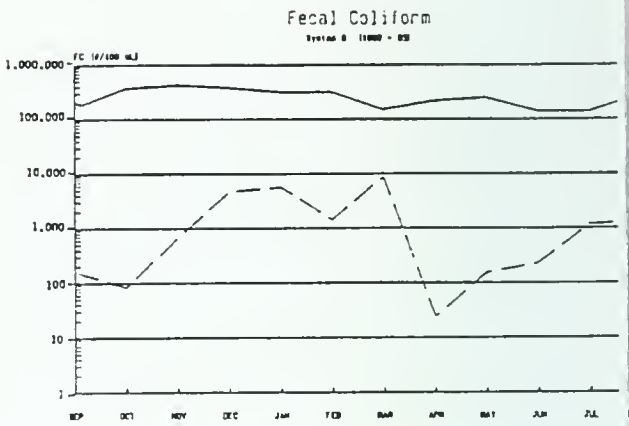
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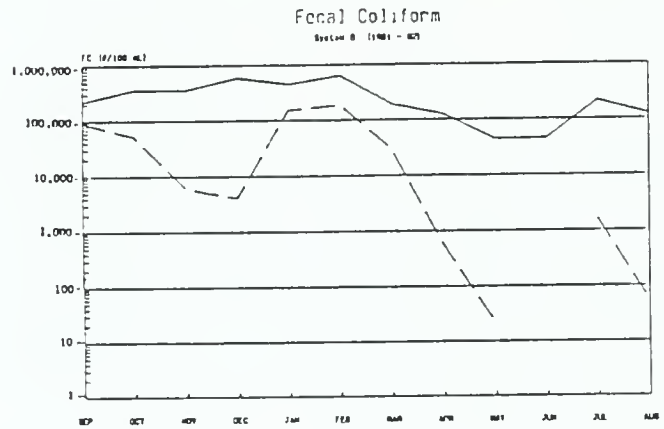
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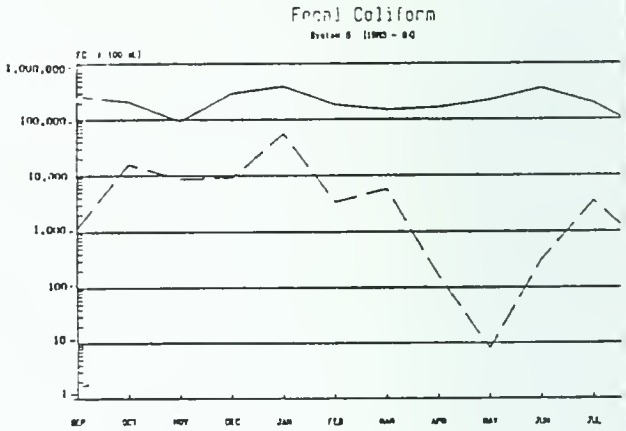
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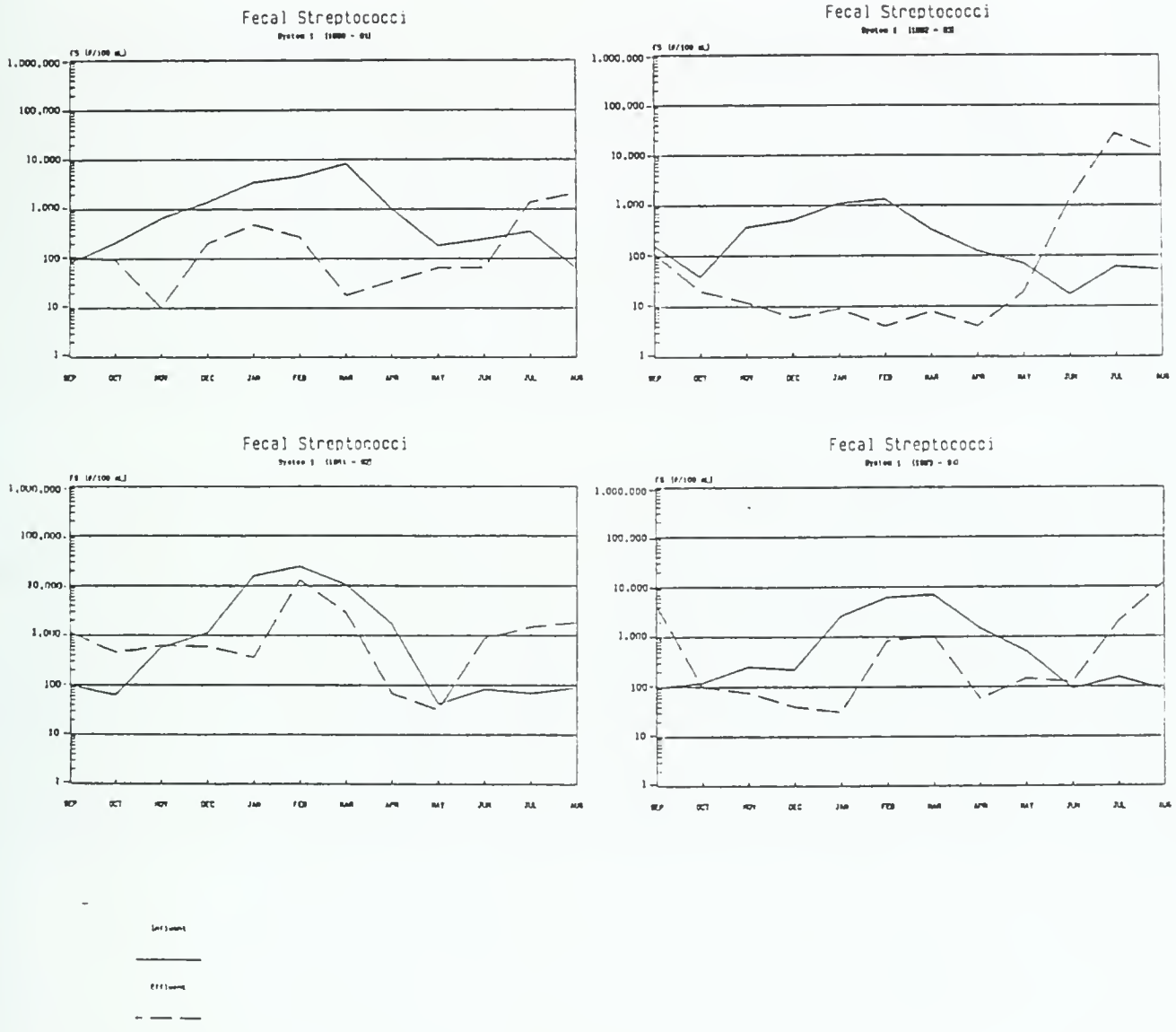
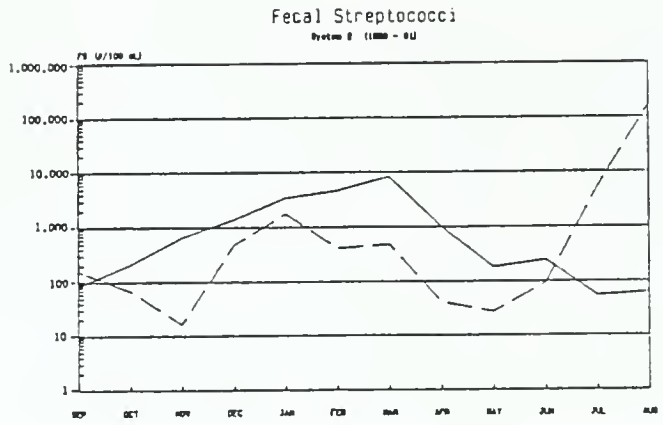


Figure A.32 Monthly geometric mean fecal streptococcus levels in Systems 1-5 (4 years).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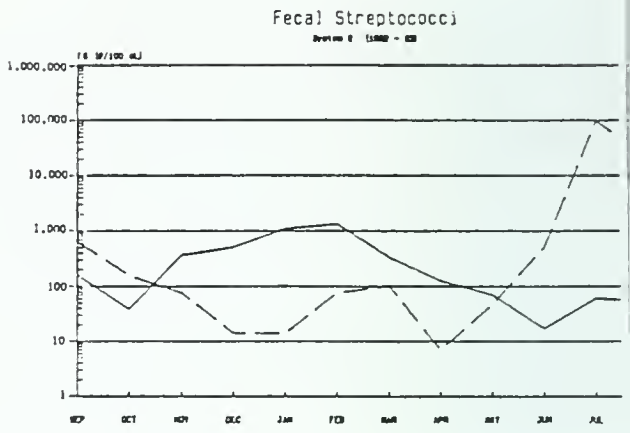
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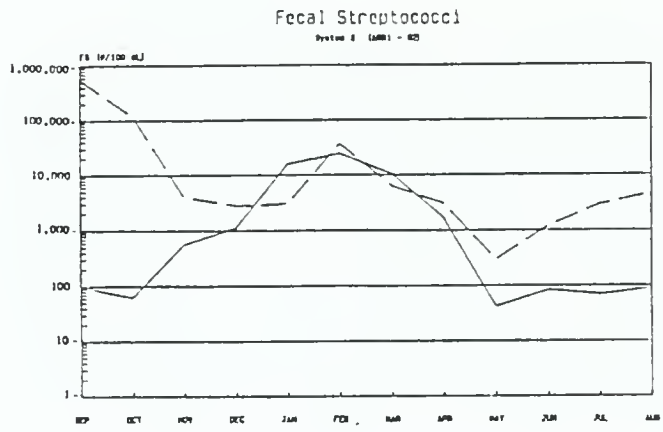
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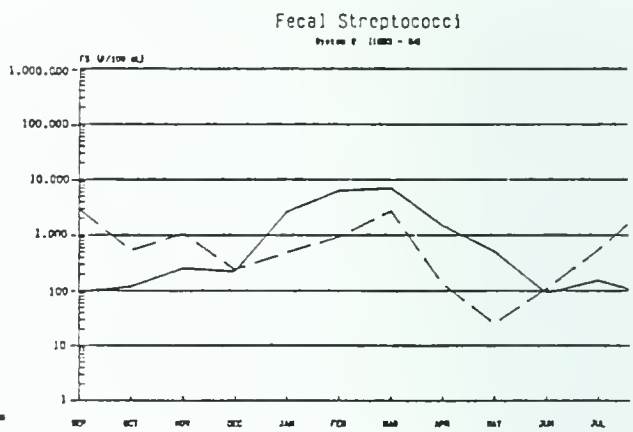
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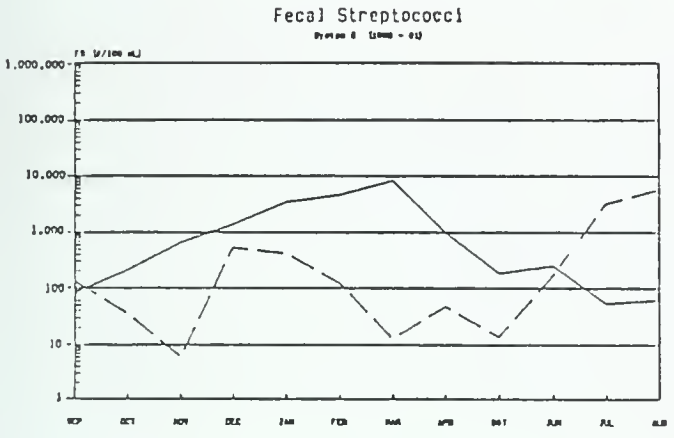
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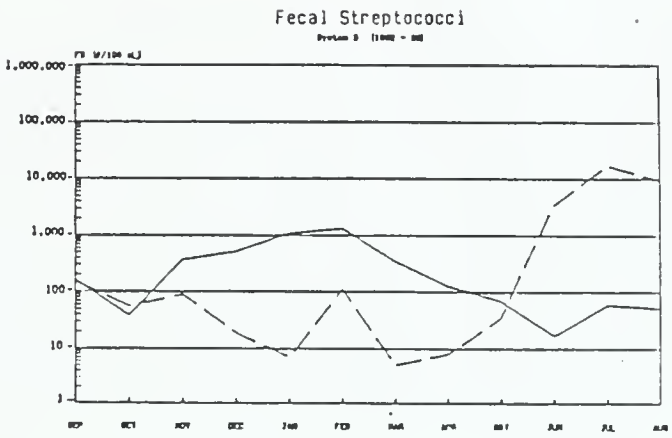
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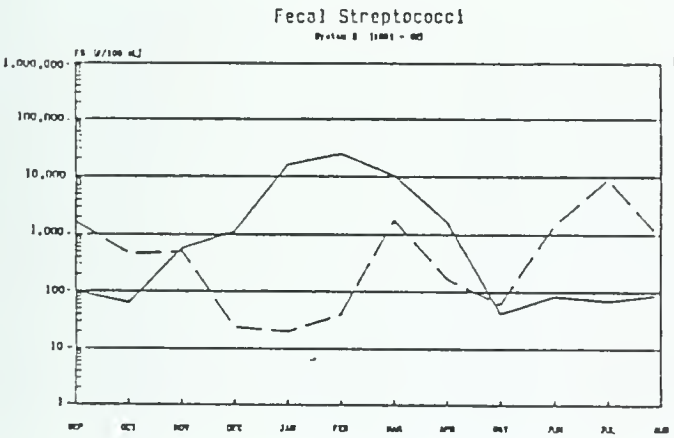
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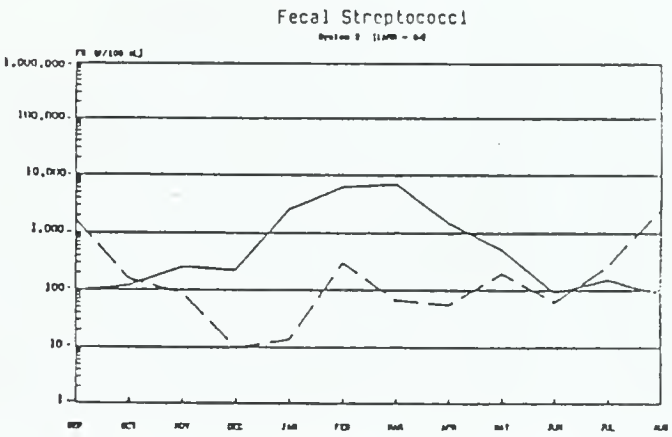
Fecal Streptococci
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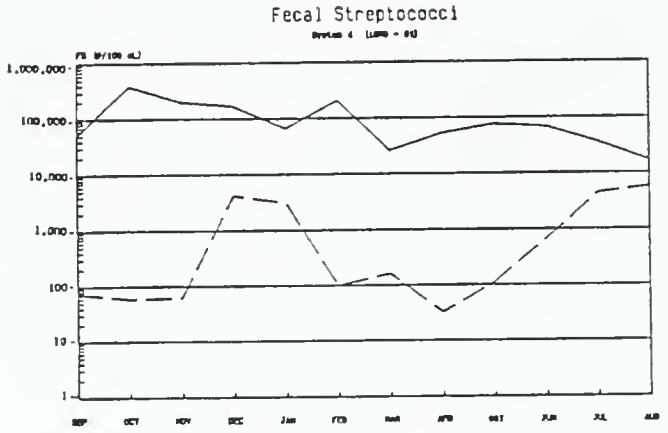
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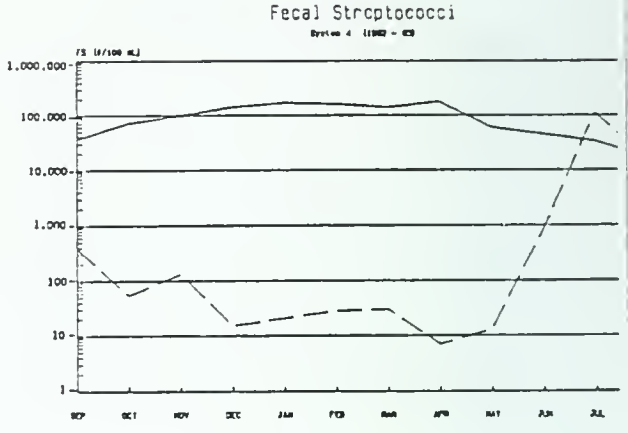
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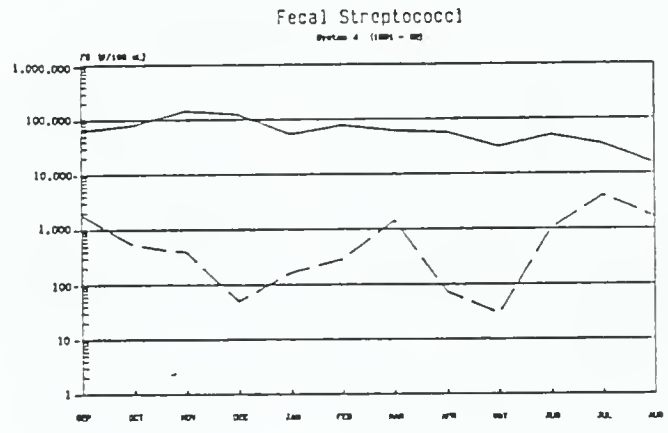
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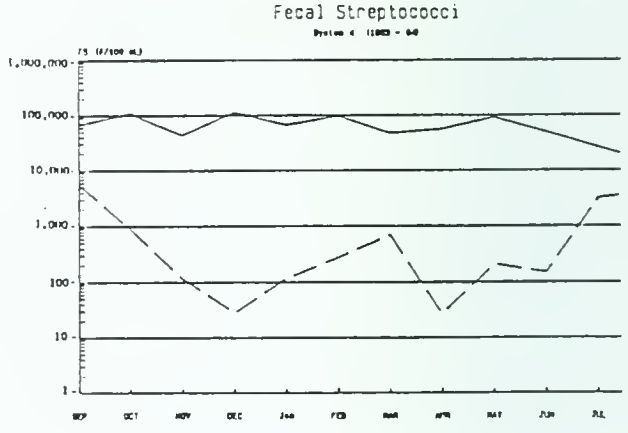
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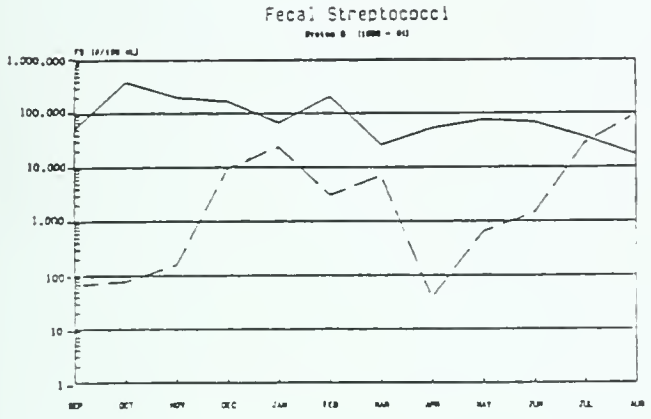
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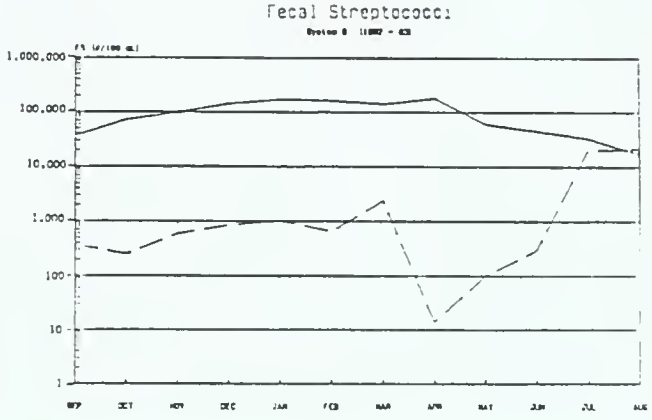
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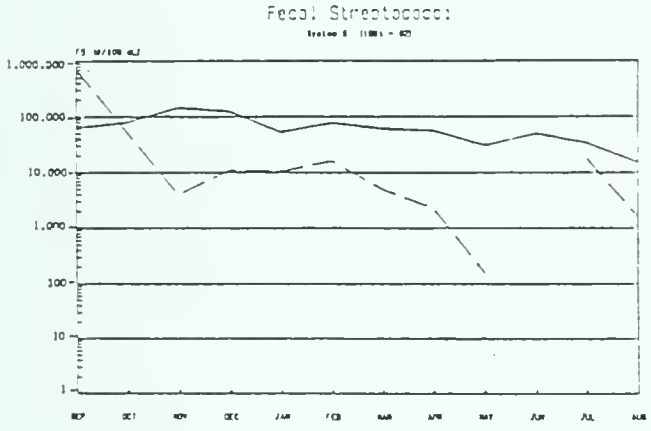
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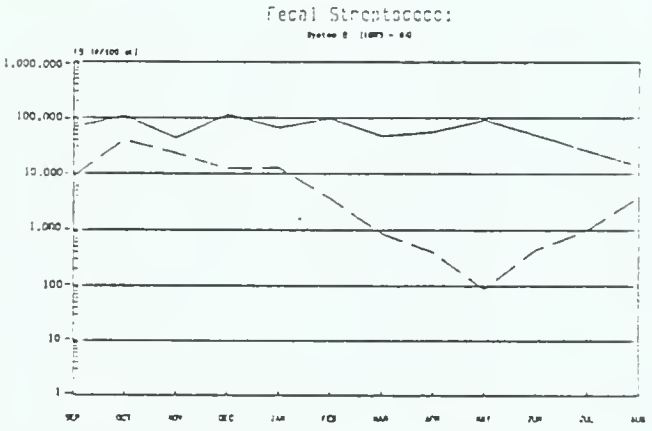
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